BENEFICIAL MANAGEMENT PRACTICES FOR THE MILK RIVER AND SOUTH SASKATCHEWAN RIVER WATERSHEDS, ALBERTA (2016 UPDATE)

A Component of the Multiple Species at Risk (MULTISAR) Conservation Strategy

Prepared for:

MULTISAR

(Alberta Conservation Association, Prairie Conservation Forum and Alberta Environment and Parks)

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Part II (Grazing Management) of this report is taken largely from the document entitled *Range Management: Training Manual for Range Managers in Alberta* prepared by Alberta Sustainable Resource Development (now Alberta Environment and Parks) (2010).

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NOTES ON THE 2016 UPDATE

A thorough review of the 2004 Beneficial Management Practices for the Milk River Basin, Alberta document was completed in 2015/16. The update document was renamed Beneficial Management Practices for the Milk River and South Saskatchewan River Watersheds, Alberta. The revised report includes new scientific literature published from 2004 to 2015. In addition, the names of the different government departments and agencies listed in this report have been updated. In several instances, species conservation status rankings had changed since 2004 and therefore had to be updated. Terminology changes were made in a few instances to provide better standardization throughout the document. Minor formatting inconsistencies were also cleaned up. Websites url's were also updated to reflect recent changes. Plant species scientific names were updated to conform to the most current listing published by Alberta Conservation Information Management System. As agreed to by the MULTISAR partners, the section on grazing management (Part II) was largely replaced with a previously developed Government of Alberta document for Crown land range managers.

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ABSTRACT

This report has been prepared as part of the Multiple Species at Risk (MULTISAR) conservation strategy for the Milk River and South Saskatchewan River watersheds in southern Alberta. This initiative was spearheaded by Alberta Environment and Parks, the Prairie Conservation Forum and the Alberta Conservation Association. The aim of the MULTISAR project is to implement a process to achieve multi-species conservation through appropriate management on critical parts of the landscape. The objectives of this project were summarized in the Year 1 Progress Report for the MULTISAR project (Alberta Species at Risk Report No. 72). At its core, the MULTISAR project emphasizes cooperative initiatives, partnerships and voluntary stewardship activities to achieve conservation of species at risk.

The Milk River Watershed is located in southern Alberta, Canada and occupies an area of approximately 6,797 km^2. The watershed extends north from the United States border along the Saskatchewan border to Cypress Hills Provincial Park and west to Whiskey Gap, Alberta (located east of Cardston). The unique landscapes, topography and remaining native prairie within the Milk River Watershed and along the North Milk and Milk Rivers and their tributaries provide habitat to a diversity of wildlife. Numerous plants and animals within the Milk River Basin are at the northern limit of their North American range and are either unique to the basin or are rare anywhere else in Alberta or Canada.

The South Saskatchewan River Watershed is situated north of the Milk River Watershed and covers an area of approximately 14,597 km^2. It occurs downstream from the confluence of the Bow and Oldman rivers near Grassy Lake, Alberta east to the Saskatchewan border. The South Saskatchewan River Watershed consists of a mix native prairie and cropland, with the dominant land-uses being ranching and farming. Important native prairie habitat in the watershed has been conserved as part of Canadian Forces Base Suffield.

The Milk River and South Saskatchewan River watersheds are located within the Grassland Natural Region and contain portions of the Dry Mixedgrass, Mixedgrass, and Foothills Fescue natural subregions (Natural Regions Committee 2006). The Dry Mixedgrass Natural Subregion occupies the majority of the two watersheds and is composed of a mix of short and mid-height grasses as well as forbs and shrubs, such as silver sagebrush (*Artemisia cana*). The Dry Mixedgrass Natural Subregion is the driest subregion in Alberta and contains important remnant prairie for species at risk.

Populations of rare or unique species within the Milk River and South Saskatchewan River watersheds are sensitive to human disturbance and incompatible land-uses that remove, fragment or lower the quality of their habitat. Although many rare and threatened species are bound by unique habitat types such as badlands or eroded sandstone cliffs and hoodoos, others are dependent on the continuation of key ecological processes such as grazing to meet their habitat needs or ecological requirements. Prior to European settlement of the prairies, a wide range of interrelated factors including drought, fire and bison grazing shaped prairie ecosystems and created dynamic and varied landscapes. Attempts to mimic or sustain these types of natural disturbances are considered necessary to maintain the structure and function of prairie ecosystems, and to accommodate the diverse needs of multiple wildlife and plant species.

The purpose of this report is to develop guidelines to assist with the design of stewardship activities within the Milk River and South Saskatchewan River watersheds as part of the MULTISAR project. An assessment of grazing processes and available management practices within the context of historic disturbance regimes and multiple species needs provides a foundation from which to design appropriate habitat conservation strategies.

The specific objectives of this report are as follows:

1. To provide an overview of historic disturbance regimes of importance in the Milk River and South Saskatchewan River watersheds (Part I).
2. To review how current grazing management practices impact or potentially serve certain ecosystem processes (Part I and II).
3. To review key grazing management principles and provide a description of seven grazing systems that are suited for use in the Milk River and South Saskatchewan River watersheds (Part II).
4. To summarize known ecological and habitat requirements for select wildlife species in the Milk River and South Saskatchewan River watersheds (Part III).
5. To evaluate range management practices for their relative value in providing habitat for select wildlife species (Part III).
6. To provide a summary of beneficial management practices (BMPs) for select wildlife species in the Milk River and South Saskatchewan River watersheds (Part III).

This document provides background information on the ecological and habitat requirements of 33 select management species within the Milk River and South Saskatchewan River watersheds. These species are considered either individually or as management groups of species with similar ecological requirements. The BMPs in Part III of this report encompass appropriate land-use and grazing management strategies to maintain or enhance habitat for these species and to avoid or minimize impacts during sensitive periods. The recommendations were derived from a literature review of the limiting factors, habitat needs and ecology of each species, knowledge of grazing management, and consultation with species and range management experts. For most species, the recommendations can be applied to the full extent of their range within the Grassland Natural Region of Alberta. A synthesis of BMPs is provided at the conclusion of Part III.

It is important to stress that the recommendations provided in Part III of this report will be subject to ongoing revision based on new information, monitoring and evaluation, and feedback from interested parties or agencies. Consequently, this report should be considered a “living document” subject to an adaptive management process of review and improvement.

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INTRODUCTION

1 PROJECT OVERVIEW

The fauna and flora of the Milk River and South Saskatchewan River watersheds have evolved in response to natural processes such as fire and native herbivore grazing as well as aboriginal manipulation of the environment. The underlying assumption of many range management practices is that techniques that best mimic local natural use patterns will be beneficial for native flora and fauna. This assumption is based on the theory that native flora and fauna are physiologically or behaviourally adapted to suit certain evolutionary disturbance regimes.

The purpose of this report is to provide guidelines to assist with the implementation of stewardship activities within the Milk River and South Saskatchewan River watersheds. This project is a component of the Multiple Species at Risk (MULTISAR) Conservation Strategy for the Milk River and South Saskatchewan River watersheds. MULTISAR is a joint initiative between Alberta Environment and Parks (AEP), Fish and Wildlife Division (FWD), Alberta Conservation Association (ACA) and Prairie Conservation Forum (PCF). A more detailed description of the objectives of the MULTISAR initiative can be found in the Year 1 Progress Report for the MULTISAR project (Quinlan *et al.* 2003).

Part I of this report provides an overview of historic natural disturbance processes and reviews how current grazing management practices impact or potentially serve certain ecosystem processes. In Part II, a description of key grazing management principles is given in addition to a description of a number of different grazing systems that are suitable for use in the Milk River and South Saskatchewan River watersheds. Part III of this report provides a description of the ecological and habitat requirements and corresponding beneficial management practices (BMPs) for 33 select management species in the Milk River and South Saskatchewan River watersheds. These species are considered either individually or as groups of species with similar ecological requirements. The main emphasis of Part III of the report is to assess the advantages and disadvantages of grazing and various grazing systems for maintaining or enhancing critical habitat for each species. The BMP recommendations developed for each species or species group address advantageous grazing practices as well as other land stewardship and habitat enhancement strategies. BMP recommendations were derived from a review of the limiting factors, habitat needs and ecology of each species, knowledge of grazing management and consultation with species and range management experts. A synthesis of recommendations is provided at the conclusion of Part III. The BMPs described in this report will be subject to on-going review and evaluation based on new information, monitoring of ongoing stewardship programs and feedback from interested parties or land management agencies.

2 STUDY AREA

The Milk River Watershed occupies an area of approximately 6,797 km^2 in southern Alberta, extending north from the United States border along the Saskatchewan border to Cypress Hills Provincial Park and west to Whiskey Gap. The North Milk and Milk Rivers flow within the watershed toward the Gulf of Mexico. The Milk River Watershed contains a diversity of landscapes, including badlands, plains, uplands and valleys. Badlands can be found in the downstream section near Lost River. Unique eroded sandstone cliffs and hoodoos characterize

many areas along the valleys of the Milk River and its tributaries. The dominant land-uses in the Milk River Watershed are ranching and farming.

The South Saskatchewan River Watershed covers an area of approximately 14,597 km^2. It occurs downstream from the confluence of the Bow and Oldman rivers east to the Saskatchewan border. The South Saskatchewan River flows north into Saskatchewan, eventually joining the North Saskatchewan River and emptying into Hudson's Bay. The South Saskatchewan River Watershed consists of a mix native prairie and cropland, both irrigated and non-irrigated. Similar to the Milk River Watershed, the dominant land-uses in the South Saskatchewan River Watershed are ranching and farming.

The Milk River and South Saskatchewan River watersheds are located within the Grassland Natural Region and contain portions of the Dry Mixedgrass, Mixedgrass, and Foothills Fescue Natural Subregions (Natural Regions Committee 2006). The Dry Mixedgrass Subregion occupies the majority of the two watersheds and is composed of shortgrass species, such as blue grama (*Bouteloua gracilis*), and mid-grasses, such as western wheat grass *(Pascopyrum smithii*), June grass (*Koeleria macrantha*) and needle grasses. The Mixedgrass Subregion is restricted to the northeast corner of the watersheds near the Cypress Hills and in the south-central area north of the Sweet Grass Hills. This subregion has a slightly moister and cooler climate than the Dry Mixedgrass Subregion, and contains similar vegetation but with a larger proportion of western porcupine grass (*Hesperostipa curtiseta*) and northern wheat grass (*Elymus lanceolatus*). The Foothills Fescue Subregion receives the greatest precipitation and makes up the smallest percentage of the total area of the two watersheds. It occupies the western part of the region, and includes the Milk River Ridge. Native grassland plant communities in the Foothills Fescue Subregion are dominated by rough fescue (*Festuca campestris*, *F. hallii*), bluebunch (Idaho) fescue (*F. idahoensis*), Parry oat grass (*Danthonia parryi*) and intermediate oat grass (*D. intermedia*).

Livestock production is the primary land-use in the Milk River and South Saskatchewan River watersheds. Three large provincial grazing reserves (*i.e.*, Pinhorn, Sage Creek and Twin River), the Onefour Research Station, as well as numerous provincial grazing leases preserve some of the native prairie within these watersheds. Oil and gas development activity (*i.e.*, exploration, drilling, *etc*.) is present throughout the watersheds. Several protected areas also occur within the study area, including Writing-on-Stone Provincial Park, a portion of Cypress Hills Interprovincial Park, Milk River Natural Area and Kennedy Coulee Ecological Reserve. A large portion of the South Saskatchewan River Watershed is protected in the Canadian Forces Base Suffield.

3 METHODS

Twenty-eight important prairie wildlife species were selected by the MULTISAR team as high priority management species for conservation in the Milk River and South Saskatchewan River watersheds (Quinlan 2004). The primary criteria for species selection included:

1. Strong representative of a group of species with similar habitat associations.

2. Strong association with a specific major ecosystem (*e.g.*, native grasslands).
3. Strong association with specific habitat structures (*e.g.*, cliffs).
4. Narrow ecological tolerances.
5. High sensitivity to habitat changes and human activities.
6. Value as a "keystone species" (*e.g.*, important prey species).

Selected species included those listed as "At Risk", "May Be At Risk" and "Sensitive" as well as priority game species and ecologically important species in the Milk River and South Saskatchewan River watersheds (see descriptions of ranks below).

During the first iteration of this report in 2003/04, a thorough literature review was conducted to compile published scientific literature and unpublished documents and reports pertaining to natural processes, grazing management and ecological and habitat requirements for the 28 select management species. Where possible, information specific to the Milk River and South Saskatchewan River watersheds was obtained. Subject searches were done primarily using the *Agricola* and *Biological Abstracts* databases. Documents and reports were also obtained from AEP libraries and websites. Professional experts in the fields of range ecology and wildlife biology were contacted to provide current information.

The Habitat Suitability Index (HSI) models that were developed as part of the MULTISAR project (Downey *et al.* 2004) helped in identifying key habitat requirements for the select management species. HSI models were prepared for 17 of the species considered in this report, including:

- Ferruginous hawk (*Buteo regalis*)
- Prairie falcon (*Falco mexicanus*)
- Sharp-tailed grouse (*Tympanuchus phasianellus jamesi*)
- Burrowing owl (*Athene cunicularia*)
- Loggerhead shrike (*Lanius ludovicianus*)
- Long-billed curlew (*Numenius americanus*)
- Sprague's pipit (*Anthus spragueii*)
- Olive-backed pocket mouse (*Perognathus fasciatus*)
- Swift fox (*Vulpes velox*)
- American badger (*Taxidea taxus taxus*)
- Richardson's ground squirrel (*Spermophilus richardsonii*)
- Prairie rattlesnake (*Crotalus viridis*)
- Short-horned lizard (*Phrynosoma hernandesi hernandesi*)
- Plains spadefoot toad (*Spea bombifrons*)
- Great Plains toad (*Anaxyrus cognatus*)
- Northern leopard frog (*Lithobates pipiens*)
- Weidemeyer's admiral butterfly (*Limenitis weidemeyerii*)

For the second iteration of this report in 2015/16, six additional wildlife species of concern were added, including:

- McCown's longspur (*Calcarius mccownii*)
- Grasshopper sparrow (*Ammodramus savannarum*)
- Chestnut-collared longspur (*Calcarius ornatus*)
- Brewer's sparrow (*Spizella breweri*)
- Eastern yellow-bellied racer (*Coluber constrictor flaviventris*)
- Canadian toad (*Anaxyrus hemiophrys*)

One other species, the western small-footed bat (*Myotis ciliolabrum ciliolabrum*), was removed from the updated report as BMPs have been developed for bats as a whole in Alberta under a separate document.

A second literature review was undertaken in winter 2015/16 focusing on scientific articles, documents, reports and books published from 2004 to 2015. The focus of the literature search was on research related to the 33 different wildlife species discussed in this report, particularly research related to habitat preferences and the effects of livestock grazing. The following databases were searched for Phase 2:

- BIOSIS Previews
- BioOne
- JSTOR Journals
- Agricola
- Academic Search Complete (EBSCO)
- Wildlife and Ecology Studies Worldwide
- Zoological Record
- Annals of the New York Academy of Sciences
- ScienceDirect (Elsevier)
- Google Scholar

Similar to Phase 1 of the project, in addition to the library databases relevant websites and online government publications were also searched for using an internet search engine. Government and academic experts were contacted for recent information pertaining to the species at risk discussed in this report. Recent updates to the MULTISAR HSI models using Resource Selection Functions (RSFs) were also reviewed and information from the new models incorporated into this report.

4 STATUS RANKS

Throughout Part III of this report, various status ranks are discussed for the different wildlife species. Ranks have been assigned to each species by different management agencies based on their perceived risk of extinction or extirpation within each respective jurisdiction. Provincially, status ranks are assigned wildlife species by Alberta Environment and Parks (AEP) using one of seven categories (Table I-1). Once a species has been designated as 'At Risk' by AEP, it undergoes a detailed status assessment and may be placed on the provincial *Wildlife Act*, usually with a ranking of 'Endangered', 'Threatened' or 'Special Concern'. Nationally, species

assessments are completed by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC), a branch of Environment and Climate Change Canada. COSEWIC uses seven slightly different categories to rank species. Wildlife species determined to be "Endangered" or "Threatened" nationally are then placed on the federal *Species at Risk Act* (*SARA*). Internationally, ranking is carried out by the International Union for Conservation of Nature and Natural Resources (IUCN). Species assessed by the IUCN are placed into one of nine categories (Table I-1).

Table I-1 Wildlife Species Status Rankings

AEP General Status[1]	**COSEWIC / *SARA* / *Wildlife Act***[2]	**IUCN**[3]
Extinct/Extirpated – Any species no longer thought to be present in Alberta (Extirpated) or no longer believed to be present anywhere in the world (Extinct).	**Extinct** – A wildlife species that no longer exists.	**Extinct –** A taxon is Extinct when there is no reasonable doubt that the last individual has died. A taxon is presumed Extinct when exhaustive surveys in known and/or expected habitat, at appropriate times (diurnal, seasonal, annual) throughout its historic range have failed to record an individual.
At Risk – Any species known to be At Risk after formal detailed status assessment and legal designation as Endangered or Threatened in Alberta.	**Extirpated** – A wildlife species no longer existing in the wild in Canada, but occurring elsewhere.	**Extinct in the Wild –** A taxon is Extinct in the Wild when it is known only to survive in cultivation, in captivity or as a naturalized population (or populations) well outside the past range. A taxon is presumed Extinct in the Wild when exhaustive surveys in known and/or expected habitat, at appropriate times (diurnal, seasonal, annual) throughout its historic range have failed to record an individual.
May be at Risk – Any species that May Be At Risk of extinction or extirpation, and is therefore a candidate for detailed risk assessment.	**Endangered** – A wildlife species facing imminent extirpation or extinction.	**Critically Endangered –** A taxon is Critically Endangered when the best available evidence indicates that it meets any of the criteria A to E for Critically Endangered, and it is therefore considered to be facing an extremely high risk of extinction in the wild.

Sensitive – Any species that is not at risk of extinction or extirpation but may require special attention or protection to prevent it from becoming at risk.	**Threatened** – A wildlife species likely to become endangered if limiting factors are not reversed.	**Endangered –** A taxon is Endangered when the best available evidence indicates that it meets any of the criteria A to E for Endangered, and it is therefore considered to be facing a very high risk of extinction in the wild.
Secure – A species that is not At Risk, May Be At Risk or Sensitive	**Special Concern** – A wildlife species that may become a threatened or an endangered species because of a combination of biological characteristics and identified threats.	**Vulnerable –** A taxon is Vulnerable when the best available evidence indicates that it meets any of the criteria A to E for Vulnerable, and it is therefore considered to be facing a high risk of extinction in the wild.
Undetermined – Any species for which insufficient information, knowledge, or data is available to reliably evaluate its general status.	**Data Deficient** – A category that applies when the available information is insufficient (a) to resolve a wildlife species' eligibility for assessment or (b) to permit an assessment of the wildlife species' risk of extinction.	**Near Threatened –** A taxon is Near Threatened when it has been evaluated against the criteria but does not qualify for Critically Endangered, Endangered or Vulnerable now, but is close to qualifying for or is likely to qualify for a threatened category in the near future.
Not Assessed – Any species that has not been examined during this exercise.	**Not at Risk** – A wildlife species that has been evaluated and found to be not at risk of extinction given the current circumstances.	**Least Concern –** A taxon is Least Concern when it has been evaluated against the criteria and does not qualify for Critically Endangered, Endangered, Vulnerable or Near Threatened. Widespread and abundant taxa are included in this category.
		Data Deficient – A taxon is Data Deficient when there is inadequate information to make a direct or indirect assessment of its risk of

		extinction based on its distribution and/or population status. A taxon in this category may be well-studied, and its biology well known, but appropriate data on abundance and/or distribution are lacking. Data Deficient is therefore not a category of threat. Listing of taxa in this category indicates that more information is required and acknowledges the possibility that future research will show that threatened classification is appropriate.
		Not Evaluated – A taxon that has not yet been evaluated against the criteria.

[1] Source: Government of Alberta (2011)
[2] Source: COSEWIC (2015)
[3] Source: IUCN Species Survival Commission (2012)

5 DEFINITIONS

Throughout this report, an effort has been made to standardize common terms. Different terms are often used in the literature to refer to the same or similar features, thereby making effective communication challenging. For example, a paddock refers to an enclosed area grazed by livestock and usually refers to the same feature as a field or a pasture. In other instances, the same terms are sometimes used to refer to different features of ecological significance for prairie wildlife species. For example, an old field could refer to a field containing native prairie or a field containing land cultivates and seeded to tame forages (*i.e.*, tame pasture). Properly distinguishing between native prairie and tame pasture can be very important in determining the habitat needs of prairie wildlife species (see Section III). An attempt was made use common terms when it was apparent what the authors were referring to. If after careful consideration it was not clear exactly what the author was referring to, the original term was retained. Common terms and their definitions are provided in Table I-2.

Table I-2 Common Ecological Terms and Definitions

Term[1]	Definition
Native Prairie (syn: Native Grassland) (incl. Native Shrubland)	Plant communities that were never cultivate (broken) and where native grass species dominate percent cover and introduced plant species are absent or have low cover. May or may not be grazed by domestic livestock (see Idle Land, below).
Tame Pasture (syn: Seeded Pasture, Permanent Cover – Grazed)	Land previously cultivated and seeded to tame forages such as alfalfa (*Medicago sativa*), smooth brome and crested wheat grass (*Agropyron cristatum*), and that is grazed by livestock.
Hay Field (syn: Permanent Cover – Hayed)	Land previously cultivated and seeded to tame forages, and that is currently used for hay production.
Cropland (syn: Cultivated Field, Fallow Field, Stubble Field)	Land previously cultivated and seeded annually to crops. Land may be cultivated annually (conventional tillage) or not (minimal tillage or no-till). Common annual crops include cultivated barley (*Hordeum vulgare*), cultivated wheat (*Triticum aestivum*) and canola/rape (*Brassica* spp.).
Idle Land	Land, generally native prairie, which is not grazed by domestic livestock.
Rangeland (incl. Native Grassland, Native Shrubland, Forest and Riparian Areas)	Land supporting native vegetation that either is grazed or that has the potential to be grazed by herbivores, and is managed as a natural ecosystem.

[1] Synonyms refer to terms that are similar with respect to their ecological function.

6 REFERENCES

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PART I:

NATURAL PROCESSES OF THE MILK RIVER AND SOUTH SASKATCHEWAN RIVER WATERSHEDS

1 OVERVIEW OF NATURAL PROCESSES

The Grasslands Natural Region of Alberta occupies approximately 95,565 km^2 of land, of which nearly 43% remains as native prairie (Prairie Conservation Forum 2000, Natural Regions Committee 2006). Three natural subregions occur in the Milk River and South Saskatchewan River watersheds: the Dry Mixedgrass, Mixedgrass and Foothills Fescue Natural Subregions. Of these, the Dry Mixedgrass Subregion occupies that largest area. The prairie landscape as we see it today has been forming for millennia. The topographic and physical characteristics have their origins in the last glaciation. The development of the landscape was influenced by the advance and retreat of several glaciations, the most recent being the Laurentide Glacier, and the erosion and deposition processes that took place following this period. The pattern and type of soils and associated vegetation that have since come to dominate the prairies is largely the result of periodic drought, flood, fire, large and small herbivore grazing and predators.

Grasslands exist as a result of natural causative agents that affect an area to favour grass or grass-like vegetation. Grasslands evolved in response to several natural agents, acting singly or, more likely, collectively. Climate, topography and wind, large ungulate herds and fire are often cited as primary natural agents favouring grasslands over woodlands. The persistence of certain vegetation or plant community types is likely the result of the ability of species/communities to adapt to a historical disturbance regime. Although the size, intensity and return interval of disturbance prior to European settlement is largely unknown, scientific and cultural knowledge indicate that natural variation was great (Bradley and Wallis 1996).

1.1 Fire

Fire is an important ecological process in terrestrial ecosystems, limiting encroachment by woody species, facilitating plant community renewal by removing excess dead plant material and recycling nutrients. Under protection from fire, areas that have historically consisted of grassland or open savanna have experienced an increase in the cover of woody vegetation (Bailey and Wroe 1974, Vogl 1974, Bock and Bock 1984). The advance of woody species in previously grassland-dominated regions is evidenced in the photographic records from near the turn of the century, historical accounts and biosequence (grassland to wooded types) of soils (Baumeister *et al.* 1996, Dormaar and Lutwick 1966).

The historic fire return interval for fescue prairie is estimated to be five to 10 years (Wright and Bailey 1982, Arno 1980). The availability and continuity of fuel, topography and climatic (mostly wind) conditions determine the propagation of fire and its behaviour on the prairie landscape (Pyne *et al.* 1996). Prairie fires would have varied in intensity and size depending on these factors. Historical accounts indicate that prairie fires often burned for days and single fires covered huge areas, running for 100 km to 200 km or more (Nelson and England 1971, Higgins 1986).

Many plant communities require fire to rejuvenate growth and return species composition to an earlier seral stage (Wright and Bailey 1982). In areas where the rate of litter accumulation exceeds decomposition, with plants curing and dying back each year, fire acts as a mechanism for accelerating the recycling of nutrients into the soil. Prairie landscapes, as influenced by

historical fire and bison grazing events, existed as a mosaic of seral communities (transitory stages of plant community development), each with unique disturbance histories.

Grasslands in the drier regions of the Great Plains are maintained by climatic factors, whereas at the fringe of grassland and forest ecotones, a combination of fire, drought and grazing/browsing serve to determine the type of vegetative cover (Coupland 1992). Since common woody species such as aspen (*Populus tremuloides*) and buckbrush (*Symphoricarpos occidentalis*) sprout vigorously following fire, occasional fire events alone would not control shrub cover. Extensive browsing of woody shoots by bison as well as wallowing, trampling and toppling of trees by rubbing against them likely contributed significantly to the suppression of aspen growth on the prairies (Campbell *et al.* 1994).

Changes in the composition of fescue prairie as a result of fire have been primarily attributed to an altered microenvironment, particularly moisture regimes (Romo 1996). Burning mostly shifts the environment in fescue prairie from one with light limitation to one that is water-limited. Grasses adapted to drier environments are favoured over species adapted to more mesic environments. Annual spring burning in the parkland causes a shift from species of the fescue prairie towards a mixedgrass prairie association (Anderson and Bailey 1980). The persistence of these drier conditions is dependent on the severity of the fire and the climatic conditions following the fire event. Under favourable climatic conditions, the recovery of burned fescue prairie to pre-burn composition and production may only require a few years (Joudannais and Bedunah 1990, Redmann *et al.* 1993).

Fire has important ecological effects on vegetation composition and structure, including productivity, insect populations and soil properties (Kerr *et al.* 1993). Fire favours vegetation that is adapted to periodic removal of above-ground growth. Fire commonly favours forbs over grasses in grasslands (Daubenmire 1968, Antos *et al.* 1983, Bailey and Anderson 1978). Plant species diversity may increase by removing litter from areas that have heavy accumulations. Excessive litter build-up can be detrimental to seeding establishment of some species. In the fescue prairie, large bunches of Foothills rough fescue (*Festuca campestris*) are more seriously damaged by fire than smaller bunches, indicating these plants are adapted to shorter intervals between fires or disturbance, which limit the expansion in plant diameter (Antos *et al.* 1983). Fescue grasslands are generally resistant to fire, as single defoliation events following fire do not have a detrimental effect on rough fescue (Bogen *et al*. 2003). This resistance is likely facilitated by reduced production in rough fescue plants after fire, which diminishes the value of grazing, especially with regard to the increased risk to the plant with subsequent grazing.

Historical records of lightning-set fires are rare compared with the accounts of Native American-set fires (Higgins 1986). Higgins (1984) found that on average six to 24 lightning fires per year per 10,000 km^2 occurred in the mixedgrass prairie during a period from 1940-1981. The majority of these fires occurred in July and August. In more mesic environments, the incidence of lightning caused fire may be less significant. In Yellowstone National Park, for example, lightning strikes on average 4 times per km^2/year, but has not initiated a single fire in the northern range despite an abundance of available fuel (Kay 1995). This is likely because when conditions are conducive to lightning strikes, the herbaceous vegetation is too green to carry a

fire. Historically, aboriginal burning occurred in every month of the year except January (Higgins 1984).

1.2 Native Americans

Fire caused by Native Americans differs from lightning fire in terms of seasonality, frequency, severity and ignition patterns (Kay 1995). Aboriginal fires were mostly set in spring, between snowmelt and green-up, or late in the fall at a time when burning conditions would not create as severe effects as those caused by lightning fires during dry periods (Kay 1995). Whereas lightning fires tend to be infrequent and intense, native burning during these periods was more frequent, but produced lower intensity fires. The impact of native burning on plant communities was undoubtedly great, contributing to the formation of the mosaic of vegetation types on the prairie that were present at the time of European settlement.

It is suggested that where precipitation is sufficient to support the growth of trees, grasslands were of anthropogenic rather than climatic origin (Denevan 1992). However, other climatic and topographical factors influence the persistence of parkland or forested vegetation. Burning at the grassland-forest transition will create drier conditions, favouring grassland and pushing back the forest edge. Some regions of the prairies may have been maintained only through nearly annual burning by Native Americans during the last 5,000 years (Anderson 1990). Native Americans were active landscape architects, using fire extensively to manipulate the plant community and distribution of game (Dormaar and Barsh 2000). Fire was used by Native North Americans to modify the plant community to maintain, for example, patches of naturally occurring medicinal and food plants such as camas (*Camassia quamash*) and wild turnip (*Psoralea esculaenta*), and many other cultural and inter-tribal relation reasons (Kay 1995). The nomadic nature of tribes influenced the occurrence of useful medicinal and food plants by the collection and cultivation of plants from certain areas and transplanting them to other areas.

Native Americans also had the capability to influence the ungulate population through hunting, which in turn would influence the native prairie. Lewis and Clark (1893) noted that “with regard to game in general, we observe that the greatest quantities of wild animals are usually found in the country lying between two nations at war.” Aboriginal hunting tended to extirpate or drive out game animals, and resource depletion around camps and villages has frequently been reported in studies of modern hunter-gatherers (Kay 1995).

1.3 Bison

Herds of bison (*Bison bison*) on the northern Great Plains distributed themselves in response to variable climatic factors, fluctuations in the quality and quantity of available forage, availability of water and hunting pressure. The number of bison in North American has been estimated at 40 to 60 million animals by Seton (1929) based on assumptions regarding carrying capacity, range area, habitats and population trends. The migratory nature of bison makes it difficult to estimate the population that existed in western Canada. From historical accounts of western Canada’s early explorers, individual bison herds would have ranged in the thousands, and millions were likely present on the grasslands of western Canada at any given time (England and DeVos 1969). Undoubtedly, their impact on the landscape was significant. Larson (1940)

suggested that their presence in the shortgrass prairie maintained that plant community in a state of disclimax and without bison grazing the community would likely be more representative of mixedgrass prairie.

Although bison grazing was prevalent throughout the prairies, their transient nature would have likely resulted in large herds not returning to these areas for several years. Bison may also have demonstrated what Epp (1988) refers to as a "dual dispersion strategy", having both migratory and non-migratory herds. River valleys, parklands, ranges of hills and sandhills with abundant water may have been inhabited by small sedentary herds of bison, which would have fed in nearby grassy uplands (Epp 1988). Bison may have also remained on the plains during mild winters (Moodie and Ray 1976).

It is generally believed that bison migration was from the mixedgrass prairie of the northern Great Plains to the fescue prairie in the foothills and parkland in Alberta and similar regions in Saskatchewan and Manitoba with the onset of winter (Moodie and Ray 1976, Morgan 1980). In early autumn soon after the summer rut, large herds of plains bison split into smaller units and migrated to wintering grounds. However, other views have challenged this notion, suggesting that bison movements were erratic, only governed by the availability of food (Hanson 1984).

Based on a preference for fescue prairie by wintering by bison, movement to the foothills and parkland was likely driven by three fundamental energy requirements to survive the winter: 1) deposit fat reserves prior to the onset of winter; 2) utilize a winter diet of adequately high energy forage; and 3) take advantage of protein-rich, early spring forage growth (Baumeister *et al.* 1996). Foothills rough fescue, the characteristic grass of the fescue prairie, initiates spring growth approximately one month earlier than dominant mixedgrass prairie species such as blue grama (*Bouteloua gracilis*). Early spring growth is typical of most cool-season grasses, and providing soil moisture is sufficient enough, additional growth may occur in the fall (Stout *et al.* 1981). This lends to the adaptation of rough fescue grassland to grazing during dormant periods. Repeated defoliation of rough fescue during the growing season can be detrimental, resulting in reduced yields, vigor and eventual elimination of plants from the community (Willms 1988, Willms and Fraser 1992). In comparison, mixedgrass communities may be more resistant to grazing during the growing season based on the historic use pattern of these ranges. Repeated defoliation at a moderated utilization level throughout the growing season generally does not negatively impact species composition in mixedgrass communities (Biondini *et al.* 1998).

Selective use of habitats or plant species by large herbivores can influence plant populations, diversity and community structure, and ecosystem processes (Vinton *et al.* 1993). The opportunity for selective grazing within plant communities in the northern Great Plains by immense migratory bison herds was likely low. Rather, the forage supply would have been completely utilized before the herd moved on as indicated by accounts in the journals of early explorers. Where vast herds of bison had passed, the ground was completely denuded of vegetation leaving little or no forage available for the explorers' horses (Nelson 1973). However, where sedentary herds existed, there would be a greater opportunity for selective grazing and regrazing of more favourable forage.

Bison behaviour and activity, besides grazing, also influenced the structure and composition of grasslands. Wallowing, pawing, trailing and other similar non-grazing bison activity creates micro-environmental effects that increase heterogeneity on the landscape. These small changes on the landscape increase the diversity of environmental conditions plants are able to take advantage of and potentially increase overall species richness (Hartnett *et al.* 1997).

Although cattle (*Bos taurus*), as large grass-feeding herbivores, may be able to fulfill the same ecological function as bison, there are inherent differences in their grazing behaviour. Cattle grazing patterns are influenced by slope, as well as horizontal and vertical distance from water, regardless of forage availability (Van Vuren 1982). Cattle use a significantly lower percentage of upland habitat compared with bison, and tend to favour floodplain habitat. Forage availability appears to be the only factor affecting bison distribution as rugged terrain seldom impedes their movement and they will travel considerable distances from water and spending less time at water sources. Bison are far more efficient water users than cattle and can better utilize lower quality, drier forage (Wuerthner 1998). The construction of fences, stock water and other developments undoubtedly alter bison and cattle foraging behaviour alike compared with the natural wanderings of large bison herds. See new articles

1.4 Other Wildlife

The Great Plains were described as teeming with abundant game in historical accounts of pre-settlement times. Nelson (1973) describes the early explorers' accounts of the variety of species such as bison, elk (*Cervus elaphus*), pronghorn (*Antilocapra americana*), wolves (*Canis lupus*) and cougars (*Felis concolor*) as common in their thousands and millions, along with a multitude of birds. The area around the Cypress Hills was likened to parts of Africa where the plains were swarming with animal life of all kinds.

Pronghorn numbers were estimated to be as numerous as the bison (Rand 1945). The historical range of pronghorn extended north to the North Saskatchewan River, east into Manitoba and west to Rocky Mountain House, Alberta (England and De Vos 1969). There is little account of the slaughter of pronghorn to the excess experienced by the bison during the same period, but their numbers decreased greatly. This may be due to a loss of suitable habitat with the extirpation of bison herds. The diet of pronghorn consists largely of forbs and browse and the abundance of suitable forage would have largely been dependent on grazing by bison herds and fire to favour these plant species. The demise of large pronghorn herds may have been linked with the extirpation of the bison as well as the construction of barbed wire fences and other obstructions to pronghorn movement with the onset of European settlement.

As a result of long-term control measures and other human activities, several species of carnivores have been extirpated from the Great Plains of North America (Jones and Manning 1996). These include species such as wolves, black bears (*Ursus americanus*), grizzly bears (*Ursus arctos*) and cougars. Conversely, species such as coyotes (*Canis latrans*) and American badgers (*Taxidea taxus*) now inhabit areas that were not used historically (Jones and Manning 1996). Carnivore predation, in addition to native hunting, may have influenced ungulate populations and distribution. According to predator-prey theory, prey populations will increase if they have a refugium where they are safe from predation (Taylor 1984). By undertaking long-

distance migrations, bison were able to out-distance most of their carnivorous and human predators (Kay 1995).

The abundance and influence of ungulate species other than bison on the prairies is questionable in some regions of western North America (Kay 1995), but the influence of the beaver (*Castor canadensis*) in shaping the landscape of western Canada is significant. Where humans wielded fire, beavers controlled the water. While beaver were commonly found along mountain streams, large numbers also inhabited watercourses on the prairies (Kay 1995). It is suggested that millions of beaver inhabited western North America before the fur trade (Johnson and Chance 1974). Beavers continually dammed up streams and rivers, often causing new watercourses to be formed. Changes in drainage patterns had distinct influence on vegetation and likely attracted other animals, such as moose (*Alces alces*) and ducks, to the area. Beaver are not nearly as common as they were prior to European settlement and are even considered ecologically extinct in regions of western North America (Kay 1995).

Small burrowing mammals may have also contributed to the structure and function of the native prairie. The black-tailed prairie dog (*Cynomys ludovicianus*) is the most widespread of the prairie dog species in the mixedgrass and shortgrass prairies of the Great Plains (Hoogland 1995). Pre-settlement distribution and abundance is largely unknown, but lands currently occupied by prairie dogs are thought to represent less than 10% of their historical range (Anderson *et al.* 1986). Prairie dogs likely inhabited the Milk River Watershed at one time, but their distribution in Canada is presently restricted to extreme southern Saskatchewan. Prairie dogs can have large and significant effects on plant productivity, community dynamics and nutrient cycling (Whicker and Detling 1988), similar, though not equal, to the effects of other burrowing mammals on the prairie. Prairie dogs are principally herbivores, and their grazing activities tend to increase the abundance of forbs in the vicinity of established colonies. These modified habitat conditions likely affected the distribution, at least on a regional level, of other foraging animals, such as pronghorn and bison (Stapp 1998). Burrowing activities that affect nutrient cycling and other associated ecosystem processes thereby may also modify soil micro-climate and plant production. The effects prairie dogs and other small mammals have on the ecosystem tended to increase the overall diversity and promote functional integrity of native grassland communities (Stapp 1998).

1.5 Implications for Ecosystem Management

Prairie species evolved in response to certain level of disturbance, and many plant communities became dependent on disturbance for their regeneration or survival (Bradley and Wallis 1996). These disturbances occurred in such a manner that promoted biodiversity while maintaining high productivity sustaining large ungulate populations. For example, more diverse plant communities are more resistant to and recover more fully from a major drought than less diverse communities (Tilman and Downing 1994). Native prairie is in a state of dynamic equilibrium, self-sustaining and resilient to disturbance within the natural range of variation. The stability of long-term primary production in prairie ecosystems is dependent on the maintenance of biodiversity (Tilman and Downing 1994).

Although it is neither feasible nor practical to manage for all components of the ecosystem, striving for ecological integrity through promotion of biodiversity and sustained function (*e.g.*, grass production) is a principle of ecosystem management. Objectives of current range management tend to be uniform distribution of use with moderate grazing pressure. These management techniques strive to maintain the health of native prairie and avoid degradation of areas where livestock may congregate, such as riparian habitats. Most grazing systems are applied with moderate grazing pressure that permit selective grazing and the creation of overgrazed and undergrazed patches. Patchy grazing contributes to landscape heterogeneity, but it is usually within fields on a small scale (*i.e.*, small patch size). Grazing systems can be used that allow the creation of planned heterogeneity on a larger scale by controlling the grazing pressure and timing of grazing within fields.

By allowing natural processes, such as erosion/deposition, drought, flood, fire and herbivory to occur on the landscape or by approximating them through management, it is assumed there will be a better chance of preserving biodiversity and sustaining ecosystem processes (Bradley and Wallis 1996). Patchy grazing on a large scale, with lightly, moderately and heavily used areas, may be desirable as these patterns of livestock use will allow for the greatest variety of plant and wildlife species. However, these practices applied on a smaller scale may not produce varying plant communities significantly large enough to be effective habitats for certain species. It may be more effective in some cases to create heterogeneous communities between fields on a landscape level. Range management that over-intensifies or homogenizes grazing consistently across the landscape will tend to reduce the range of natural variation. Specialized grazing systems can create a mosaic of grass cover types to satisfy a diversity of animal species, while still accommodating livestock grazing.

Prior to agricultural settlement, a wide range of interrelated factors such as drought, fire and bison grazing created dynamic and varied landscapes. This likely included extremes in environmental conditions from highly impacted areas (due to grazing, fire or drought, for example) to areas of low use. These extremes are largely absent from the present day landscape, but may have been very important to vegetation and wildlife dynamics. A better understanding of missing features and natural processes on ecosystem health may be required for prairie conservation.

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PART II:
GRAZING MANAGEMENT

The following section is taken largely from the document entitled Range Management: Training Manual for Range Managers in Alberta *prepared by Alberta Sustainable Resource Development (now Alberta Environment and Parks) (2010).*

1 BACKGROUND

1.1 Principles

Range management is referred to as the art and science of producing sustained yields for livestock and wildlife while maintaining ecosystem integrity for a variety of purposes (Society for Range Management 1998). Proper range management leads to increased livestock production and improved watershed and ecosystem stability (Holechek *et al.* 2003).

There are four key principles of range management:

1. **Stocking** – Balance livestock demand with the available forage supply. Ensure there is sufficient carryover to maintain other rangeland values (*e.g.*, watershed protection) by stocking at or below the grazing capacity.
2. **Distribution** – Control livestock distribution and access to minimize selective grazing behaviour and prevent re-grazing of plants. Cross-fencing to implement a rotational grazing system (see below), when combined with water development and salting practices, concentrates livestock and forces more uniform livestock distribution while grazing.
3. **Season** – Graze range at the right time of year. If possible, avoid grazing native prairie during the vulnerable spring growth period. The process of alternating the sequence in which the pastures are grazed from year to year minimizes the negative effects of spring grazing (*i.e.*, periodic deferral of grazing during the most sensitive plant growth stage).
4. **Rest** – Allow each field a period of rest from grazing during the active growth season to manage and maintain the vegetation. Cross-fencing of the range also allows part of the range to be grazed while other areas are allowed to rest and recover in the absence of grazing.

1.2 Range Health

Range health is tool for assessing the ecological health of a particular upland site or plant community. Range health builds on the traditional range condition approach that considers plant community type in relation to site potential, but adds new and important indicators of natural processes and functions. Range health is measured by comparing the functioning of ecological processes of an area of rangeland to a standard known as an ecological site description. An ecological site is similar to the concept of range site, but includes a broader list of characteristics. An ecological site, as defined by the Task Group on Unity in Concepts and Terminology (1995), "is a distinctive kind of land with specific physical characteristics that differs from other kinds of land in its ability to produce a distinctive kind and amount of vegetation."

Range health rates a plant community's ability to perform certain key ecological functions, including (Adams *et al.* 2010):

- Net primary production.

- Maintenance of soil/site stability.
- Capture and beneficial release of water.
- Nutrient and energy cycling.
- Plant species diversity.

Healthy plant communities are productive, stable and diverse. Unhealthy communities are unstable, less diverse and susceptible to erosion and lost productivity. Healthy native grassland communities have: high cover of native grass species, especially those expected to occur on the particular ecological site; good structural diversity; high litter cover; low amounts of bare soil and erosion; and no noxious weeds.

Range health can be assessed in grassland, forest and tame pasture plant communities. The following criteria are used to evaluate the health of grassland and forest communities (Adams *et al.* 2010):

- Ecological status (*i.e.*, similarity of the plant community to the climax or reference plant community for the ecological site).
- Structural heterogeneity and the diversity of vegetation layers present.
- Comparison of litter (grassland)/LFH (forest) cover to expected levels in lightly-grazed communities.
- Percentage of the site affected by soil erosion, including evidence of soil movement and the percent cover of human-caused bare soil.
- Cover and distribution of noxious weeds.

The following criteria are used to evaluate the health of tame pasture communities (Adams *et al.* 2010):

- Canopy cover of all introduced species combined (not including cover of native species, nuisance or noxious weeds and woody plants).
- Canopy cover of productive native and introduced forage species.
- Canopy cover of weedy or disturbance-induced species;
- Litter cover and distribution.
- Percent of site covered in human-caused bare soil or affected by erosion.
- Percent cover and distribution of noxious weeds.
- Percent cover and distribution of woody re-growth.

Grassland communities can also be described as ‘modified’ based on the degree of alteration from ‘normal’. Modified grassland plant communities are those in which more than 70% of the plant composition is comprised of non-native species. These communities were never broken (cultivated), but through heavy disturbance or natural invasion they have become dominated by exotic plant species. Modified communities are unlikely to revert back to their native state and are given a lower allowable score for ecological status than native communities (Adams *et al.* 2010). The ecological status of modified communities is rated according to the percent composition of desirable and productive non-native species with good vigour versus the percent composition of weedy and disturbance-induced non-native species.

Range health assessment sites receive a score out of 100 according to how well they rate for each of the five ecological indicators. Scores are then broken into one of three categories: Healthy, Healthy with Problems or Unhealthy (Figure II-1; Table II-1).

Table II-1. Range Health Categories and Percent Ranges

Health Category	Range Health Percent Range	Riparian Health Percent Range	Functioning	Comments
Healthy	75 to 100	80 to 100	Proper Functioning Condition	Plant community is performing key ecological functions.
Healthy with Problems	50 to 74	60 to 79	Functional at Risk	Many key ecological functions are still being performed, but there are signs of stress to the system. System may be vulnerable to erosion.
Unhealthy	0 to 49	0 to 59	Non-functional	Most key ecological functions are not being performed or are seriously impaired. Resiliency and stability of the plant community has been compromised.

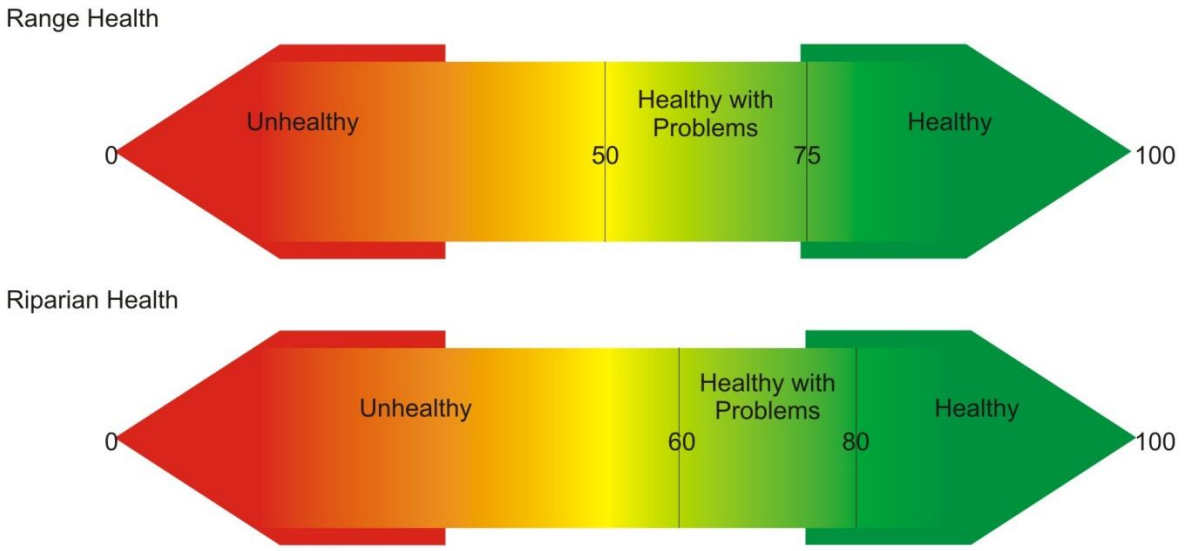

Figure II-1. Range and Riparian Health Scores and Classes

1.3 Riparian Health

Riparian areas are the transitional areas that occur between upland and aquatic habitats along the margins of lakes, wetlands, creeks and rivers. Riparian areas generally have one or more of the following characteristics: (1) wetland hydrology; (2) hydric soils; and/or (3) hydrophytic vegetation (Thompson and Hansen 2002). Riparian areas are often quite lush and tend to remain greener longer in the growing season than upland areas. Despite occupying a small proportion of the landscape, riparian areas are disproportionately important in terms of their value to both

humans and wildlife (Fitch *et al.* 2003) and are known hotspots of biodiversity (Naiman *et al*. 1993). They also play an important role in flood protection, maintenance of water quality and provision of wildlife habitat (Adamus and Stockwell 1983).

Riparian health is similar to range health except that it is specifically tailored to the riparian environment. Riparian health assessments evaluate if riparian areas are performing certain key ecological functions, such as trapping sediment, filtering water, building and stabilizing banks and storing water and energy (Fitch *et al.* 2001). Similar to range health, riparian health assessment sites are compared to known benchmarks to rate ecological functioning. Riparian health assessments can be used to assess both lotic (*i.e.*, flowing water) and lentic (*i.e.*, still water) systems.

The following indicators are used to assess the health of lotic riparian communities (Fitch *et al.* 2001):

- Vegetation cover.
- Cover and distribution of invasive weed species.
- Cover of disturbance-increaser species.
- Preferred tree and shrub establishment and regeneration.
- Utilization/browsing intensity of preferred trees and shrubs.
- Woody vegetation removal other than browsing.
- Cover of dead and decadent woody vegetation.
- Streambank root mass protection.
- Physical alteration of the streambank due to human activity.
- Physical alteration of the floodplain due to human activity.
- Amount of human-caused bare ground.
- Degree of channel incisement.

The following indicators are used to assess the health of lentic riparian communities (Ambrose *et al.* 2004):

- Vegetation cover.
- Cover and distribution of invasive weed species.
- Cover of disturbance-increaser species.
- Preferred tree and shrub establishment and regeneration.
- Utilization/browsing intensity of preferred trees and shrubs.
- Woody vegetation removal other than browsing.
- Human alteration of vegetation.
- Physical alteration due to human activity (including severity).
- Amount of human-caused bare ground.
- Degree of artificial water withdrawal.

Similar to range health, riparian health is scored out of 100, with scores falling into one of three broad classes (Figure II-1; Table II-1). Healthy riparian areas generally have high vegetation cover; low amounts of invasive and disturbance plant species; a good mix of young and old

woody species; and little physical impact and low amounts of bare soil as a result of human and livestock activity, among other characteristics.

1.4 Livestock Grazing, Wildlife and Range/Riparian Health

Range and riparian health are analogous to the suitability of grassland, forest and riparian habitats for different wildlife species. Range and riparian health rate the degree to which grazing has impacted plant communities and measure the corresponding level of ecological impairment (if any). While some wildlife species are more generalist in nature or show little preference for heavily grazed or lightly grazed habitats, many species show preference for one plant community type over the other (Bock *et al.* 1993, Knopf 1996). For example, heavily grazed sites with short vegetation are often preferred by mountain plovers (*Charadrius montanus*), whereas species such as Sprague's pipits (*Anthus spragueii*) favour lightly grazed native grassland with little alteration (Alberta Sustainable Resource Development [SRD] 2003, Sliwinski 2011). Similarly, heavy grazing in riparian areas may negatively impact certain wildlife species, such as prairie rattlesnakes (*Crotalus viridis*) (Gardiner *et al.* 2013). Research suggests that while range health itself may not be the greatest predictor of, for example, grassland bird abundance in the prairies, the variables it measures, such as litter mass, bare ground cover and vegetation volume, are good predictors (Henderson and Davis 2014). Therefore, range and riparian health may be useful indicators not only of the intensity of livestock grazing and how well the plant community as a whole is functioning, but also the suitability of that community for certain wildlife species.

Direct impacts of livestock grazing on wildlife depend largely on stocking rates, climate and soil (Fulbright and Ortega-Santos 2006). Uniform grazing as a result of high stock density can reduce the quality of wildlife habitat by decreasing plant species diversity and the escape, resting, screening and thermal cover wildlife need to survive. Additional factors that may contribute to livestock grazing impacts on wildlife include the effects of fencing and water developments, disease transmission, compatibility of livestock grazing with predators, and the economic value of livestock grazing versus wildlife.

Properly managed livestock grazing can improve wildlife habitat by increasing species diversity. However, when used as a habitat management tool livestock grazing is very wildlife species-specific in its benefits. Soil fertility and rainfall are important factors influencing the effect of herbivores on plant species diversity. In habitats with relatively high grass biomass and non-limiting precipitation, grazing by livestock may increase plant species richness. However, when grazing increases from moderate to heavy, plant species richness declines.

Livestock stocking rates resulting in about 25% or less removal of annual herbaceous production generally are not detrimental to wildlife habitat on North American rangelands (Galt *et al.* 2000; Lyons and Wright 2003). Light grazing (*i.e.*, less than 35% use of primary forage species) to moderate grazing (*i.e.*, 35% to 45% use of primary forage species) usually encourages forb production, which may benefit certain species at risk, such as pronghorn (*Antilocapra americana*) and greater sage-grouse (*Centrocercus urophasianus*) (Mitchell and Smoliak 1971, Barnett and Crawford 1994).

Livestock grazing is most likely to have a positive effect in areas with more than 50 cm (20 in.) of annual rainfall. In drier areas, properly managed light to moderate grazing usually does not

damage wildlife habitat, but it is unlikely to improve it (Lyons and Wright 2003). Removal of 50% of the annual production of forage by livestock generally does not result in rangeland deterioration in humid bioclimatic zones, but stocking rates resulting in consumption of ≥50% of the forage standing crop in arid and semiarid bioclimatic zones commonly result in overgrazing, habitat degradation and increased competition with wildlife (Fulbright and Ortega-Santos 2006).

Conservation of native prairie is vital for the survival of species at risk in Alberta. Many native wildlife species, including species at risk such as ferruginous hawks (*Buteo regalis*) and burrowing owls (*Athene cunicularia*), are dependent on native prairie for all or portions of their life cycles (Schmutz 1987, Clayton and Schmutz 1999). Cultivation or breaking of native prairie is thought to contribute to the declines seen in recent years of many species at risk (*e.g.*, Cotterill 1997, Lauzon 1999). The dominant land-use on native prairie in southern Alberta is domestic livestock grazing. If properly managed, livestock grazing is compatible with conservation of species at risk (*e.g.*, Lusk 2009, Ranellucci *et al.* 2012). Moreover, while some species at risk (*e.g.*, long-billed curlews; Prescott and Bilyk 1996) will make use of altered habitats such as tame pastures, these species are generally more productive in native prairie. In addition, use of tame pastures by some species is dependent on the amount of native prairie in surrounding areas (Davis *et al.* 2013). Therefore, conserving the remaining native prairie in southern Alberta is not only compatible with a traditional lifestyle (*i.e.*, ranching); it is also compatible with the conservation of species at risk.

1.5 Establishing Management Objectives

Management objectives that are compatible with the needs of the landowner, the resources and the long-term viability of species biodiversity and wildlife habitat must be determined. Before any management system can be implemented, current range and riparian health as well as the desired plant community must be identified. Knowledge of the range resource base assists in focusing and formulating realistic management goals and facilitates the implementation of an effective management plan. Defining management objectives makes it possible to develop strategies directing the desired change in the soil-plant-animal complex. The desired plant community may achieve a number of management objectives within the criteria of maintaining a healthy ecosystem, conserving biodiversity, promoting water and soil conservation and providing an adequate amount and quality of forage for livestock.

2 STOCKING RATES

The principal consideration of any rangeland grazing system is to balance livestock needs with the available forage supply through proper stocking rates. The stocking rate is the number of animals on a defined area during a given period of time. The proper stocking rate, also called the ecologically sustainable stocking rate (ESSR), is the number of animals that may safely graze a defined area without degrading the soil or vegetation, and which allows range in poor health to improve. This balancing act is referred to as proper use.

Proper use considers how much of the forage produced during the growing season may be grazed. Proper use factors are designed to ensure that sufficient biomass carryover is allocated for maintenance of ecological functions (*e.g.*, soil protection, nutrient cycling, hydrological

function) and plant community services (*e.g.*, wildlife forage use, habitat maintenance) (ASRD 2004). In this approach, the allowable portion that can be grazed varies depending on the natural subregion, plant growth stage, climate, ecological site and plant community type. For example:

- In mixedgrass prairie, the recommended proper use factor is commonly 30% to 40%.
- In prairie sandhills communities, the allowable amount is 30% because of the possible damage from trampling and litter removal.
- In more mesic grassland (*e.g.*, fescue prairie), proper use may be as high as 50%.
- In tame pastures, proper use varies from 40% to 70%.
- Forest communities may only sustain 25% utilization of the total understory without causing undesirable changes in plant community structure or species composition. Higher levels of use may cause site degradation.

In addition, different plant species in these communities will have varying tolerance levels due to plant morphology and growth patterns. The ungrazed portion of a plant community, called carryover, is left for protective cover. Carryover serves to maintain root vigour, protect soil, conserve scarce moisture and provide emergency forage. If livestock distribution is not a problem, reducing stocking rates improves range health more than any other grazing strategy.

Alberta Environment and Parks (AEP) has developed a set of range plant community guides for most natural subregions in Alberta (*e.g.*, Adams *et al*. 2013). These guidebooks include detailed information on the types of plant communities known to occur in each subregion. In addition, detailed plant species compositions and ESSRs are provided for each plant community. When all communities in a given field are combined, an ESSR can be determined for the field as a whole. Detailed range inventories allow for accurate plant community mapping, assessments of ecological health and development of ESSRs to benefit livestock as well as wildlife species at risk.

3 GRAZING SYSTEMS

3.1 Background

Grazing systems are designed to manipulate livestock in a planned manner, optimizing livestock and forage production while maintaining the ecological integrity of the range through correct stocking rates and forage use levels. An effective grazing system controls the timing, intensity and frequency of grazing of individual plants and improves livestock distribution. Selection of an appropriate grazing system is dependent on vegetation type, physiology of the plants being grazed, type of livestock and objectives of the manager. Implementation of an appropriate grazing system, when combined with ESSRs, is helpful in maintaining or improving range and riparian health of native rangeland. Grazing systems can be used by range managers to address specific resource management needs for wildlife, fisheries, water management, forestry and others.

Livestock are creatures of habit and they will not willingly distribute themselves uniformly across the rangeland. However, they can be trained to adapt. Grazing systems place a series of constraints on grazing animals to ensure better distribution. However, grazing systems are not effective if used in isolation (Holechek *et al.* 2003). Overgrazing as a result of high stocking

rates is the major factor causing degradation of rangelands. Therefore, when used without due respect to ESSRs, grazing systems alone will not maintain or improve range and riparian health.

Grazing systems should always be applied with clear insight as to the desired objectives and what is to be achieved, and with adequate flexibility to adjust to changing conditions. Grazing systems should never be rigid in application. Adaptation is always critical in biological systems where variability is normal. When developing a grazing strategy, range managers must always consider how the natural system functions. Grazing systems that make use of different areas of the available rangeland at different times of the year may complement or compete with various wildlife species. Management of plant communities depends on an understanding of the ecological processes and ecology of the communities being managed for.

Grazing systems should attempt to balance livestock needs with the needs of the range ecosystem. They must resolve a basic dilemma: in spring, livestock producers are short of grass, but grazing during this season is most harmful to native range plants and soil. However, with many areas now having more grazing-resistant naturalized plant species, grazing systems can be designed so naturalized grasses are grazed in early spring, and grazing of native grasses can be deferred until later in the growing season. A well-designed grazing system can achieve optimum livestock production and, at the same time, maintain or improve rangeland while it is being grazed. Most grazing systems focus on improving animal gain per acre by maximizing forage production and optimizing its use. However, focusing strictly on animal gains may be detrimental to the forage plants on which the animals depend. A well-designed grazing system manages livestock grazing to provide adequate periods of rest and recovery so that grazed plants regain vigour, set seed and store food reserves in their roots and stems.

The use of one or more grazing system within a ranch operation or grazing lease must consider a number of factors, including:

- **Topography and vegetation:** The more complex the topography and vegetation patterns, the more difficult it is to manage rangeland and range livestock efficiently. It is essential to adapt the grazing system to the needs of each property or lease.
- **Climate and weather:** The climate affects which types of rangeland ecosystems occur in an area by influencing water supply, soil quality and plant communities. The weather is a yearly phenomenon that affects forage quantity and quality as well as livestock performance. Rainfall and temperature from May to July affect how much forage is available for the current year as well as subsequent years.
- **Use of tame grasses:** Tame pastures provide good early and late season forage if managed properly and allow for deferred use of native prairie fields. Skim grazing of tame grasses within predominantly native rangeland may also be a desired management strategy in some areas.
- **Riparian areas:** Riparian areas are highly desirable grazing locations for livestock due to the abundance of forage and ready supply of water. Therefore, riparian areas often receive high use by livestock if access to these areas is not managed properly.

Each livestock operation must develop a grazing management system tailored to its resources and objectives. Managers have the option to combine traits from more than one grazing system

to develop a system that suits their needs and goals. There is no universal best grazing system applicable to native prairie and some systems are only successful in certain environments. The implementation of a specialized grazing system does not ensure good range and riparian health. The principles of range management must still be adhered to. The success of a grazing system relies on proper stocking rates, animal distribution, proper use, monitoring and adaptation.

3.2 Types of Grazing Systems

Many different types of grazing systems have been developed for native prairie, each having distinct characteristics, objectives, advantages and disadvantages. The two major categories of grazing systems are continuous and rotational. Within these categories there are many possible variations.

3.2.1 Continuous Grazing System

Continuous grazing, also called season-long grazing, is a traditional grazing system whereby livestock are left to graze in a single large field for the entire grazing season. Continuous grazing was very popular with pioneer settlers and remains popular with ranchers to this day.

Advantages and disadvantages of continuous grazing are outlined below:

Advantages:

- On less productive ranges where drought is frequent, continuous grazing can be successfully applied.
- Some ranchers in the mixedgrass prairie of southeastern Alberta successfully use a modified continuous grazing system. Productive, healthy range can be maintained there when managers limit grazing to moderate forage utilization rates and moderate stocking rates. This may require the use of additional water developments, keeping salt well away from water to encourage better livestock distribution and riding to move livestock into lightly grazed areas (*i.e.*, secondary range).
- Continuous grazing is convenient, requires little fencing and labour, is inexpensive and requires minimal management input.
- Continuous grazing is a very good approach when patch diversity is desired for wildlife habitat objectives.

Disadvantages:

- Continuous grazing leads to patches of undergrazed range away from water and patches of overgrazed range near water.
- Continuous grazing is not recommended for higher productivity Alberta rangeland because fields soon become patchworks of overgrazed areas favoured by livestock and ungrazed areas where forage is underutilized.
- Continuous grazing can lead to loss of desirable forage species. Cattle are selective grazers, and as such they naturally favour certain plants and areas of a field, which they will graze and re-graze. Thus, favoured plants – the palatable, productive, deep-rooted species – are soon replaced by low-producing, or unpalatable and weedy, grazing-

resistant plants. In addition, the soil beneath these heavily grazed tends to lose its organic matter and fertility over time, leading to soil degradation.

- Continuous grazing with a heavy stocking rate is particularly harmful to native vegetation and soils.

3.2.2 Rotational Grazing Systems

Rotational grazing systems rely on patterns of deferral and rest that help to sustain and improve rangeland health. Deferral means to delay grazing until a critical growth stage of the plant has passed (*e.g.*, seed set). Such deferral is intended to permit seed production, seedling establishment and restoration of plant vigour. Deferral, along with moderate stocking rates, promotes the full growth potential of range vegetation. Usually, the first field of native rangeland to be deferred in spring is the one that was grazed first in the previous year. Once the first field is grazed in early spring, there is a long rest period of no grazing to enable plant re-growth and energy replenishment. In the complementary grazing system, tame grasses are grazed in early spring while grazing of native grasses is deferred. Certain tame grasses, including crested wheat grass (*Agropyron cristatum*), smooth brome (*Bromus inermis* ssp. *inermis*), timothy (*Phleum pratense*) and Kentucky bluegrass (*Poa pratensis* ssp. *pratensis*), should be grazed each year in spring when they are most palatable. Some of these fields will also benefit from a modified rotational grazing system. Rest means to restrict an area from grazing for an entire year or growing season to allow forage plants to grow, recover and set seed.

There are many types of rotational grazing systems. The three examples presented below are some of the more popular ones being used today. Deferred rotation, complementary rotation, rotational seasonal grazing, twice-over rotation and high intensity-low frequency (HILF) grazing systems are currently in use in Alberta.

Deferred-Rotation Grazing

Deferred-rotation grazing involves alternating the deferral period between fields. The simplest form of deferred-rotation is the two-field switchback system, whereby the first field grazed in a given year is alternated between two fields. With more fields, the early grazing period is cycled amongst other fields from year to year, and the harmful effects of early season use are minimized. Deferred-rotational grazing is a popular and widely used grazing system. Often three, four, six or eight fields are grazed in rotation with a different field being grazing each year during the vulnerable spring growing period.

Advantages and disadvantages of deferred-rotation grazing are outlined below:

Advantages:

- Deferral of grazing in early spring is especially important for sustaining the health of many native range plants. Range plants rely on food reserves in their roots and stems to fuel new growth. For this reason, early season grazing, especially at heavy rates, can be particularly harmful to range plants. Grazing during early spring will cause the plant to start re-growth from depleted plant energy reserves. Deferral provides an extra period of growth to restore depleted reserves. An old rule-of-thumb from Agriculture and Agri-food Canada's early range research suggests that every day of delayed grazing in spring translates to two or three extra days of grazing at the end of the season.

- Higher stocking rates may be possible in a deferred-rotation system if cross-fencing is used to create smaller, more uniform fields. These more uniform fields will reduce selective grazing and improve livestock distribution because cattle will graze areas that previously went ungrazed.
- Rest after first grazing enables plants to regrow before being grazed again.
- Deferred-rotation grazing is normally recommended for grassland range types in the absence of tame pasture for spring grazing.
- Deferred-rotation grazing is recommended for use in rangelands that have significant amounts of riparian (wetland) areas because cattle prefer to congregate in those areas, leading to trampling, bare ground and erosion. Most fields with a large amount of riparian land should normally not be grazed in early spring due to the risk of compaction of saturated soils.

Disadvantages:

- Compared to continuous grazing, deferred rotation grazing usually requires more capital costs, such as fencing and water supply development.
- Deferred grazing usually requires more managerial and labour costs, including planning, range and livestock management, riding for better livestock distribution and use and time spent monitoring grass growth.
- Rotational grazing does not offer advantages over continuous grazing in terms of either increased vegetation production or increased animal production.

Complementary Grazing System

Complementary grazing is a specific type of deferred-rotation grazing system. It involves the use of tame pasture and native rangeland in a manner that benefits both types of plant communities as well as livestock. The system employs spring grazing of tame forages followed by summer and fall grazing of native range. The tame forages utilized are selected based on their adaptation to spring use. Complementary grazing may also involve using tame pastures to extend the fall grazing period. Russian wild rye (*Psathyrostachys juncea*) Altai wild rye (*Leymus angustus*), red fescue (*Festuca rubra*) and fall rye (*Secale cereale*) have all been used successfully for this purpose. A high level of nutrition can be maintained to the end of the grazing season.

Advantages and disadvantages of complementary grazing are outlined below.

Advantages:

- Complementary grazing systems may be appropriate for rangelands throughout Alberta. For example, in the Dry Mixedgrass Natural Subregion, crested wheat grass has supplied spring forage for many years.
- Many producers can adopt complementary grazing by simply applying this strategy to existing pasture units.
- The strategy incorporates permanent deferral for native range during the vulnerable spring growth period.

Disadvantages:

- If a grazing operation does not already include some tame pasture, then implementing a complementary grazing system would require breaking and seeding native prairie, which adds costs, carries with it the risk of failed forage establishment and reduces the value of the land for many wildlife species at risk. If it is necessary to create some tame pasture, it is best to convert marginal cropland.
- Complementary grazing also involves other capital costs as well. Experience shows that tame pasture and native grassland should be fenced into separate fields for effective management. In particular, fencing is needed to control stocking rates, timing and duration of grazing in each range type.

Rest-Rotation Grazing System

Rest-rotation grazing differs from the deferred-rotation system by enabling one field in the grazing management unit to be rested from grazing for a 12-month period while the remaining fields are grazed at their appropriate ESSRs. The next year, a different field is rested for a whole year. The rest-rotation grazing system was implemented originally as a replacement of the continuous (season-long) grazing system. Rest-rotation grazing systems are typically implemented using four fields. While in some rest-rotation designs the other fields absorb the extra grazing load for that year, this should be avoided if possible as any gains in improved range and riparian health may be lost.

Advantages and disadvantages of rest rotation grazing are outlined below:

Advantages:

- Rest-rotation grazing may be useful in some situations as a range health improvement practice provided that stocking rates in the remaining fields remain at sustainable levels.
- Rest-rotation grazing allows one field to be completely rested from grazing, enabling forage plants to grow and set seed. This provides a seed source for range regeneration.
- Rest-rotation allows rhizomatous forage plants to expand and establish in bare areas.

Disadvantages:

- Holechek *et al.* (2003) argued that “the problem with rest-rotation is that the benefits from rest may be nullified by the extra use that occurs on the grazed pastures.” Therefore, light to moderate stocking rates must be maintained, which might translate to a reduction in stocking rates.
- In Alberta, many of the normal objectives of rest-rotation grazing can be achieved with deferred-rotation due to improved moisture conditions. Alberta rangelands are adapted to moderate levels of grazing use, and spring or summer deferment of overgrazed areas will permit these plant communities to gradually recover range health without the total absence of livestock grazing for 12 months.

3.2.3 Other Grazing Strategies

Intensive Grazing

Most intensive grazing systems are based on the Savory grazing method (Savory 1988), and are commonly referred to as HILF grazing, short-duration grazing and time-controlled grazing. All these systems follow the general concept of very high stocking rates and utilization followed by long periods of rest. High stock densities increase competition for forage between animals, forcing each to spend more time eating and less time wandering. Competition also forces animals to be less selective when grazing. Livestock will eat plant species that would generally be ignored in other grazing systems. This may result in a reduction of less desirable plant species in the field. The system enables more rigid control of animal distribution with the use of numerous smaller grazing units, and is therefore similar to rotational grazing. The intensive grazing concept was introduced with the objectives of improving the chemical and physical properties of the soil and promoting grassland succession (Savory 1983).

HILF grazing is based on high stocking densities that force the animal to use the available vegetation. Relatively long grazing periods are then followed by long recovery periods. HILF grazing is used most successfully in regions characterized by high rainfall and long growing seasons (Fraser 1993).

Short-duration grazing involves relatively high stock densities coupled with short grazing periods. Grazing periods are fixed according to the estimated time needed by key forage species to recover from grazing events. The number of days of grazing in a field is determined by the number of fields available to use in the system and the recovery time required for the particular plant community the grazing system is applied in.

Time-controlled grazing is similar to short-duration grazing in that stocking densities are high, but this system recognizes that both recovery grazing times vary with the growth rate of key forage species. Grazing periods are short (*i.e.*, one to three days) during rapid growth and longer (*i.e.*, seven to 14 days) during periods of slow growth or dormancy (Abouguendia and Dill 1993). Recovery times vary from 14 to 90 days for this type of system depending on forage growth (Fraser 1993). Time-controlled grazing requires an understanding of the time needed for plant recovery.

Skim Grazing

Skim grazing is not usually considered a grazing system, but is useful to mention as a grazing strategy to be used in conjunction with a particular grazing system. It is similar to complementary grazing in terms of utilizing exotic (non-native) and native species during their season of best use. However, it does not require entire fields of seeded tame forages. Skim grazing is used to promote livestock use of non-native grasses that have invaded native rangeland. The goal is to gain additional by forage by grazing tame forages during their season of best use. Secondary effects may include reducing seed production and potentially limiting the spread of these exotic species in native rangeland, although these effects have not been experimentally demonstrated.

When utilizing this skim grazing, fields should be grazed for a short duration in the spring to avoid the utilization of native species. Foothills rough fescue (*Festuca campestris*), if present in the field, may be preferentially grazed by livestock early in the growing season (Moisey *et al.* 2003, 2005), thereby reducing the vigour of this important native grass species. The grazing preference for Foothills rough fescue over exotic grasses at this time is dependent on the amount of standing litter within the rough fescue plant.

3.3 Riparian Area Grazing

Riparian areas and riparian health were discussed in Section 1.3. In terms of livestock grazing, riparian areas are some of the most challenging plant communities to manage. Livestock often prefer to remain in riparian areas because water, forage and shelter are in close proximity within a small area, and because these areas are a source of water and green forage long after upland plant communities have dried out. The result then often becomes degradation of these important plant communities and erosion and pollution of our vitally important watercourses and waterbodies.

Continuous (season-long) livestock grazing in riparian areas is no longer an acceptable range management practice. Range managers need to plan grazing systems that simulate the natural systems that enabled riparian vegetation to not only survive but to thrive. The following management practices can be effective in achieving successful riparian area management, including the recovery of riparian health (Fitch *et al.* 2003):

- **Alter livestock distribution.** Placing watering sites, salt, minerals, winter feeding and rubbing posts away from riparian areas, and using specialized herding techniques to move livestock to upland areas, are important management techniques to maintain riparian health.
- **Control access to riparian areas.** Fencing off riparian fields enables them to be managed separately from adjacent upland communities. Fencing, and particularly electric fencing, is an effective method of ensuring livestock are only allowed into riparian areas during periods when the plant communities are at low risk from livestock damage. Corridor fencing can be used to permit occasional light grazing use during non-critical periods.
- **Alter the timing of grazing.** Spring is not the ideal time to graze riparian areas as the banks and shores of watercourses and waterbodies are often saturated with water and susceptible to compaction. Similarly, later summer and fall grazing of riparian areas can result in damage to woody vegetation. Grazing for short periods in mid-summer may result in the least amount of damage to riparian areas.
- **Add longer rest periods to the grazing cycle**. A longer rest period during the growing season enables riparian plants to re-grow and recover following defoliation.
- **Manage grazing intensity.** Overgrazed and vulnerable riparian vegetation will recover more rapidly with several years of lighter grazing use (*i.e.*, about 25% to 40% use of the forage) followed by a rest period.
- **Use an appropriate grazing system.** A grazing system that incorporates riparian fields can be implemented to enhance livestock production and to maintain higher rangeland and riparian plant community health and productivity.

3.4 Summary

A grazing system is a range management tool that addresses current problems like poor range health, and focuses on future goals such as sustaining or improving forage production, obtaining optimum stocking rates and providing for sustainable livestock productivity and also sustainable non-grazing values. A grazing system helps livestock producers to control when and where their livestock graze. It provides a method of balancing livestock production with the current carrying capacity of the range resource.

There is no "best" grazing system. Each grazing lease and ranch is unique. Grazing systems must be planned and adapted to each unique operation to meet individual production goals and resource maintenance requirements. Most important, grazing systems must be based on an understanding of range ecosystems and sound range management principles. On larger properties, several grazing systems may be more appropriate than a single one. Where there are substantial differences in elevation and range plant community types on one ranch or grazing lease, the grazing system selected will need to be modified and adapted to the unique physical and biological attributes present. Designing the appropriate system will depend on both the management goals of the operation as well as the maintenance requirements of the rangeland resource.

Planning, testing, trial and error, careful observation and ongoing adjustments in response to weather, forage conditions and range and riparian health are necessary in utilizing grazing systems effectively. Grazing systems do not replace an experienced range manager's ability to evaluate and respond to the effects of localized overgrazing, drought, snow, slow plant growth during cold springs, poisonous plants, special needs of livestock and other factors. Experienced, knowledgeable managers can react appropriately to difficult circumstances to the benefit of rangeland, livestock and other resource interests.

4 GRAZING DISTRIBUTION

Even when proper stocking rates have been established, the selective grazing habits and patterns of livestock may result in localized overgrazing. Many complex factors contribute to distribution problems, but chief among them are topography, water locations and vegetation. Under minimal management, cattle prefer to graze, drink and loaf in valley bottoms or flat areas close to water while avoiding steep slopes, unpalatable vegetation, biting insects and thick brush. It takes much planning to change such habits. Contemporary grazing systems are one of the tools to more efficiently use rangelands (see discussion above). Regardless of the grazing system chosen, however, livestock still may not utilize the range resource in the desired fashion. To that end, a number of tools are at the range manager's disposal to improve livestock distribution. These include:

- **Water:** The distribution, quality, quantity and seasonal availability of water are key parts of each management system. Livestock watering developments are among the most effective tools to improve livestock distribution.
- **Fencing:** Subdividing large fields into smaller ones with fencing allows for greater livestock manipulation. Locating fences such that they follow natural topographic

features is often recommended. Fencing out riparian areas is now a recognized strategy to maintain the health of these vitally important plant communities.

- **Salting:** Placing salt and minerals away from water sources promotes more uniform grazing across a field. Salt/minerals may be placed as much as 1 km to 2 km away from water.
- **Herding:** Herding can be used to move livestock into areas with poor accessibility, leading to more uniform grazing use or use of underutilized areas.
- **Culling:** Ranchers can also choose to remove animals with poor distribution habits, such as those cattle that prefer remain along valley bottoms.
- **Trail development:** Building trails through natural barriers such as dense forest and rougher topography can facilitate better livestock movement patterns and more uniform use of a field.

5 SEASON OF USE

Range plants native to Alberta generally are most vulnerable to heavy and sustained grazing use during the approximately six to eight weeks of spring. Range grasses are actively growing then, and the plant energy reserves are depleted to the lowest point in the annual growth cycle. The low level of these reserves makes plants very vulnerable to damage from overgrazing. In some plant species there is a secondary low period in the energy cycle during flowering and seed-set in mid-summer. Fall and winter is the period when plants are dormant, and grazing or browsing usually has the least negative effect. Dormancy is also the period when energy reserves are at their annual peak.

Range plants rely on adequate root and leaf tissue to attain maximum forage production. Once the plant starts growing, its growth potential will depend on leaf tissue that remains ungrazed. Early grazing that removes too much leaf area will impose a heavy energy drain on range plants. However, early use for calving where numbers, duration and utilization are closely monitored may not be harmful. Persistent, heavy, season-long grazing that begins in early spring and ends in the fall will cause desirable range plants to decline in vigour and eventually die.

Protecting native plant communities during vulnerable periods, when combined with other practices, allow range managers to achieve optimum forage yields. This protection may be accomplished by grazing systems such as complementary and deferred-rotation that provide periods of deferral until a sensitive growth phase of a plant is past or when the range is ‘ready’.

Range readiness may be gauged by the occurrence of a specific growth stage of one or more of the key forage plants (*e.g.*, flowering, seed ripe, *etc.*). The concept of ‘range readiness’ is controversial because the timing varies from year to year due to weather conditions. In one year, unusually warm weather in early spring may favour early plant growth, while in the next spring range readiness may be delayed due to cold temperatures and little precipitation. Range readiness also varies with plant community and location. In a given area, grasslands usually are ready for grazing before forest understories. Trees in the forest shade the soil so it remains cooler than in adjacent grasslands. Often, the forest is not ready for grazing until about two weeks after the adjacent grassland. Within a forest, range readiness is often one to two weeks ahead in grass-

covered linear rights-of way and early succession clear-cuts, due to the higher spring soil temperatures in the more open land than in the heavily shaded forest. In hilly or mountainous areas and in coulees, south-facing slopes are ready for grazing about two to three weeks ahead of north-facing slopes where soils are colder and growth is slower. Southern Alberta mixedgrass prairie rangelands are generally ready for grazing several weeks ahead of rangelands in central and northern Alberta.

6 REST

Rest is an important strategy in any grazing management plan to help maintain or restore plant vigour and range and riparian health. Grazing during the growing season when plants are actively growing, repeated over consecutive years, is known to be harmful to plants (Romo 2006). After being grazed, plants need time to recover. How much time depends on a number of factors, such as the type of plant species; when grazing occurred and the growth stage of the plant when grazed; how closely the plant was grazed; growing conditions; and initial plant conditions (Romo 2006). Managers must provide effective rest periods when there is sufficient moisture and time to produce adequate root and leaf re-growth, and replenish both plant energy reserves and meristems. Like grazing deferral, effective rest is normally provided through a planned grazing system. A single grazing period per year is normally recommended in Alberta for prairie rangelands given the relatively short growing season.

Some of the benefits of providing rest for native grasses include (Romo 2006):

- **Increased forage production.** Fields that receive adequate rest produce more forage more consistently over time.
- **Improved vigour.** Native grasses receiving adequate rest are more vigorous than those that have been repeatedly grazed.
- **Better drought tolerance.** Well-rested native grass-dominated fields are more drought-tolerant than overgrazed rangeland.
- **Reduced susceptibility to weed invasion.** Native rangeland that receives adequate rest is more resistant to noxious weed invasion than overgrazed native grassland.

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PART III:

BENEFICIAL MANAGEMENT PRACTICES FOR SELECT MANAGEMENT SPECIES IN THE MILK RIVER AND SOUTH SASKATCHEWAN RIVER WATERSHEDS

I: BIRDS

A. RAPTOR GROUP (FERRUGINOUS HAWK, SWAINSON'S HAWK, GOLDEN EAGLE, SHORT-EARED OWL AND PRAIRIE FALCON)

1 INTRODUCTION

The purpose of this report is to summarize and compare the ecology and habitat requirements of five raptor species found within the Milk River and South Saskatchewan River watersheds: the ferruginous hawk (*Buteo regalis*), Swainson's hawk (*Buteo swainsoni*), golden eagle (*Aquila chrysaetos*), short-eared owl (*Asio flammeus*) and prairie falcon (*Falco mexicanus*). Based on this information, the potential effects of grazing and various grazing systems on raptors and their prey are discussed. This discussion is followed by a summary of recommended beneficial management practices to enhance raptor habitat in the Milk River and South Saskatchewan River watersheds. These recommendations can be applied to the range of these species within the Grassland Natural Region of Alberta (Natural Regions Committee [NRC] 2006). Lastly, a brief summary of additional research needs is presented.

As predators, raptors have a vital role to play in the prairie ecosystem. In Alberta, the *Wildlife Act* and *Migratory Birds Convention Act* afford protection to raptors from being killed or harassed by people. However, it is impossible to effectively protect prairie raptors without appropriate management of their prey and habitat (Paton 2002). The majority of the raptors discussed in this report share similar foraging and/or breeding habitat requirements, including a reliance to a lesser or greater degree on native prairie where livestock grazing is the dominant land-use. Consideration of the effects of grazing on raptors is therefore highly relevant. It is also important to consider strategies to minimize potential impacts due to other human activities, such as industrial development, which can pose a risk to these species or their habitats.

2 FERRUGINOUS HAWK

2.1 Background

The ferruginous hawk is the largest hawk in North America (Semenchuk 1992). The species has both light and dark phase individuals, of which the former are more common. Light phase ferruginous hawks have white bodies with reddish-brown shoulders, backs and rumps (Semenchuk 1992). The ferruginous hawk breeds in 17 states in the United States and three Prairie Provinces in Canada (Schmutz 1999). A grassland species, approximately 12% of its breeding range lies within Canada (Schmutz 1999, Committee on the Status of Endangered Wildlife in Canada [COSEWIC] 2008a). In Alberta, ferruginous hawks breed throughout much of the southern portion of the province, with few breeding pairs found north of Consort (Schmutz 1999). High densities of ferruginous hawks and ferruginous hawk nests have been found along the upper portions of the Milk and St. Mary rivers (Erickson 2000, Quinlan *et al.* 2003). The majority of ferruginous hawks that breed in Alberta spend the winter in Texas (Schmutz and Fyfe 1987). In general, the winter range of ferruginous hawks extends throughout much of the southwestern United States into Mexico (Schmutz 1999).

The ferruginous hawk is considered 'At Risk' in Alberta under the General Status listing (Government of Alberta [GoA] 2016) and is designated as 'Endangered' under the provincial *Wildlife Act*. Nationally, ferruginous hawks have been designated as 'Threatened' by COSEWIC and are listed as such under Schedule 1 of the *Species at Risk Act* (*SARA*) (COSEWIC 2008a, 2016). Internationally, the ferruginous hawk has been designated as a species of 'Least Concern' by the International Union for Conservation of Nature and Natural Resources (BirdLife International 2012a).

Range-wide declines in ferruginous hawk populations are suspected to have occurred over the past few decades (Schmutz 1999). Historic records indicate at least a 40% reduction in the ferruginous hawk breeding distribution in Alberta (Schmutz 1999). Populations in Alberta increased from 1982 to 1987, were similar between 1987 and 1992 and decreased from 1992 to 2000 (Taylor 2003). In 2008, the Alberta population was estimated at 618 (+/- 162) breeding pairs (COSEWIC 2008a). This number increased to 865 (+/- 201) pairs in 2015; however, the overall population trend has been declining since 1982 (Redman 2016).

Habitat alteration and fragmentation due to cultivation and intensified industrial land-use, fire suppression, variable food availability, availability of nesting sites, human disturbance during nesting, vehicle collisions, and poisoning of primary prey species are all factors contributing to the decline of ferruginous hawks in Alberta and throughout its range (Schmutz 1999, SRD and ACA 2006).

A Recovery Plan for the ferruginous hawk was released by the provincial government in 2009, covering the years 2009 to 2014 (Alberta Ferruginous Hawk Recovery Team [AFHRT] 2009). The plan discusses limiting factors and ongoing threats, describes essential habitat needs, and discusses an action plan, including habitat management and reduction of human disturbance and mortality, to promote recovery of the ferruginous hawk in Alberta.

2.2 Ecology

Ferruginous hawks are usually monogamous and will have one mate for one to several breeding seasons (Schmutz 1999). Hawks will pair up prior to or soon after arriving on the nesting grounds in Alberta in late March to early April (Schmutz 1999). Nest building starts in April and females lay one to five eggs in April or early May (Schmutz and Hungle 1989). Ferruginous hawks construct nests on the ground or in elevated structures such as trees (see Section 2.3.3). Male hawks provide food for the young while female hawks are primarily responsible for incubation. In Alberta, hatching occurs in late May to early June and has been correlated with the emergence of young Richardson's ground squirrels (*Spermophilus richardsonii*) (Schmutz *et al.* 1980). Based on a study of 629 nests in North Dakota, Gilmer and Stewart (1983) found that hawk pairs produced an average clutch of 4.1 ± 1.0 eggs. Schmutz *et al.* (2008) observed an average clutch size of 3.2 (n = 7,129) and average brood sizes of 2.71 near Hanna, Alberta and 2.79 near the Kindersley-Alsask area of Saskatchewan.

Ferruginous hawk juvenile mortality (during the first year) is estimated at 65% (Schmutz and Fyfe 1987), but is thought to decline to approximately 25% among adults (Woffinden and Murphy 1989). Fledglings and adults typically remain near the nest for one month or move short

distances to favoured hunting areas within their territories (Schmutz 1999). Ensign (1983) noted that juveniles remained in the vicinity of the nest during the first week after fledging, making flights of up to 200 m. Flight was generally observed to be mastered in the second week. Juvenile hawks begin their southerly migrations in August, while adults have been known to remain in the nesting area as late as mid-October (Bechard and Schmutz 1995).

2.2.1 Diet

Small mammals, typically *Leporidae* (rabbits and hares) and *Sciuridae* (ground squirrels and prairie dogs), make up the majority of the breeding season diet of the ferruginous hawk. In the mixedgrass prairie of Alberta and North and South Dakota, Richardson's ground squirrels are the primary prey species consumed (Lokemoen and Duebbert 1976, Gilmer and Stewart 1983, Schmutz 1987). Ground squirrels can constitute up to 89% of prey items consumed during the nestling period in Alberta (Schmutz *et al.* 1980). Ferruginous hawk pairs have been estimated to consume approximately 400 to 480 ground squirrels per breeding season to maintain the pair and its young (Michener 1997). Schmutz (1989) noted that ferruginous hawk breeding density near Hanna, Alberta compared similarly to the abundance of Richardson's ground squirrels. Gilmer and Stewart (1983) also suggested that nest densities and reproductive success of ferruginous hawks in their North Dakota study area was likely dependent on the abundance of Richardson's ground squirrels. Other important prey species include black-tailed jackrabbits (*Lepus californicus*), white-tailed jackrabbits (*L. townsendii*), northern pocket gopher (*Thomomys talpoides*) and meadow voles (*Microtus pennsylvanicus*) (Lokemoen and Duebbert 1976, Ensign 1983, Gilmer and Stewart 1983). Black-tailed jackrabbits comprised between 79% and 89% (by weight) of ferruginous hawk prey items in two years of study in northern Utah and southeastern Idaho (Howard and Wolfe 1976). Birds, such as the western meadowlark (*Sturnella neglecta*), and insects typically constitute a minor component of ferruginous hawk diets (Dechant *et al.* 2002a). Olendorff (1993) suggested that ferruginous hawks can switch to other prey when their principal prey species declines, where alternative prey species are available.

In years of low prey availability, ferruginous hawk nest success and clutch size declines significantly (Ensign 1983). Woffinden and Murphy (1989) and Ensign (1983) both report higher nestling mortality in years of low prey numbers.

2.2.2 Predators

There are limited documented cases of predation on ferruginous hawk nests in the literature. It has been suggested that ground nests are more susceptible to predation by coyotes (*Canis latrans*), American badgers (*Taxidea taxus*) and foxes (*Vulpes* spp.), whereas aerie are vulnerable to American crows (*Corvus brachyrhynchos*), common ravens (*Corvus corax*), great horned owls (*Bubo virginianus*) and golden eagles (Bechard and Schmutz 1995). COSEWIC (2008) has stated that adult ferruginous hawks are mainly targeted by great horned owls and humans. Predation is not considered a major problem for the species (Travsky and Beauvais 2005); however, increasing habitat alterations may lead to increased predator populations and interactions in the future (Commission for Economic Cooperation 2005).

2.3 Habitat Requirements

2.3.1 General

Ferruginous hawks are birds of open areas, and occur primarily in the Mixedgrass, Dry Mixedgrass, Northern Fescue and Foothills Fescue subregions of Alberta's Grassland Natural Region as well as sporadically in the Central Parkland Subregion of the Parkland Natural Region (Schmutz 1999, NRC 2006). Uncultivated grassland forms a major component of ferruginous hawk habitat (Olendorff 1973, Gilmer and Stewart 1983, Konrad and Gilmer 1986, Schmutz 1987). In general, ferruginous hawks breed where grazing is the dominant land-use (Schmutz 1999). Sparse riparian forests, periphery forests, terrain features such as cliffs and rock outcrops, as well as isolated trees and small groves, add to the suitability of an area for ferruginous hawks (Olendorff 1993).

As part of the Multiple Species at Risk (MULTISAR) conservation program in southern Alberta, a Habitat Suitability Index (HSI) model was developed for ferruginous hawk habitat use in the Milk River Watershed (Taylor 2004a). Using information on known locations as well as the biology and ecology of wildlife species, HSI models use various physical and habitat variables to predict where suitable habitat for the species may be found on the landscape. The HSI model developed for the ferruginous hawk was based on the following assumptions: native prairie provides higher quality habitat due to less disturbance; areas of less disturbance are selected for nesting; breeding areas overlap or are adjacent to foraging areas; and foraging areas are suitable for Richardson's ground squirrel, the ferruginous hawk's main prey item (Taylor 2004a). Two variables were included in the final HSI model: native prairie cover and soil texture. Habitats with higher native prairie cover and with moderately coarse textured soils have a higher suitability rating as these soils are required by most burrowing mammals.

Also as part of the MULTISAR program, a second HSI model was developed for ferruginous hawk habitat use in the headwaters of the Oldman River Watershed (Taylor 2004b). Similar variables and criteria were used as the aforementioned HSI model, with the addition of two new variables: natural subregion and tree cover. The Grassland Natural Region and its subregions were given higher importance than the Parkland or Rocky Mountain natural regions, and habitats with less than 15% tree/shrub cover were given preference over habitats with higher woody species cover.

A third HSI model was developed by MULTISAR in 2009 to better refine ferruginous hawk habitat use. This model included up to three variables. High quality ferruginous hawk habitat was assumed to occur in the same location as Richardson's ground squirrel habitat; near known locations of historic ferruginous hawk nests; or near steep valley and coulee slopes, including cliffs (*i.e.*, the Grassland Vegetation Inventory [GVI] Badlands/Bedrock ecological site; Alberta Sustainable Resource Development 2010).

2.3.2 Breeding Habitat and Land-use

For all raptors, breeding habitat is comprised of nesting habitat and suitable foraging habitat. In general, suitable ferruginous hawk breeding habitat consists of at least 50% native prairie with

solitary or small groups of trees at least 500 m away from disturbances (Schmutz 1982, Dechant *et al.* 2002a).

Ferruginous hawk nesting density has been negatively correlated with the amount of cultivation present (Konrad and Gilmer 1986, Schmutz 1987, Downey 2005). As the amount of cultivation increases, foraging opportunities for ferruginous hawks decline while the potential for human disturbance during breeding increases. Ferruginous hawks are particularly easily disturbed during the early stages of nesting (Dechant *et al.* 2002a). In South Dakota, nests further than 2.47 km from occupied buildings had an 11.4% greater probability of fledging young than nests closer to buildings (Blair 1978). In Alberta, Schmutz (1982) noted that the majority of ferruginous hawk nests were located further than 500 m away from active farmyards. In North Dakota, ferruginous hawks avoided nesting in cropland or within 700 m of occupied buildings (Gaines 1985). Lokemoen and Duebbert (1976) found that 11 of 12 ground nests were located further from human activity than randomly selected points in mixedgrass prairie in South Dakota. Ensign (1983) commented that nest locations in southeastern Montana indicated a selection for sites removed from areas of vehicular traffic and/or constant human activity.

2.3.3 Nesting Habitat

Ferruginous hawks take advantage of natural and artificial structures for nesting, where available, provided an adequate forage base is available in the vicinity (Dechant *et al.* 2002a). Cliff ledges and the tops of hoodoos along the upper Milk River provide important nesting habitat for ferruginous hawks within the Milk River Watershed (Quinlan *et al.* 2003). Ferruginous hawks will also commonly nest in trees, man-made structures (*e.g.*, nest poles, roofs of abandoned building or artificial platforms) or steep slopes (Schmutz 1987).

Olendorff (1993) analyzed data from 2,119 ferruginous hawk nests and found that trees or large shrubs (49%) were used most commonly, followed by cliffs (21%), utility structures (12%) and dirt outcrops (10%). Ground nests made up only 6%, haystacks 2% and buildings 0.1% of the total nests sampled. Ground nesting was likely more prominent prior to European modification of the prairies through fire suppression and shelterbelt planting (Schmutz 1999). In areas where trees are sparse, ground nests are more common (Dechant *et al.* 2002a). For example, Ensign (1983) found that of 91 nests found in rangelands in southeastern Montana, 97% were on the ground. Ground nests are typically located on cutbanks of varying steepness (Schmutz 1982), on hill tops, ridges or along the upper slopes of southerly to westerly aspects (Lokemoen and Duebbert 1976, Blair 1978, Ensign 1983, Gilmer and Stewart 1983). Lokemoen and Duebbert (1976) speculate that nests are oriented to allow incubating birds to rise easily into prevailing winds. In central North Dakota, the majority of ground nests were found on top or next to a large boulder or outcrop (Gilmer and Stewart 1983). Nest sites that offer optimal vantage points are usually selected (Ensign 1983). In southeastern Montana, occupied ground nests were found on slopes between 15% and 30% (Ensign 1983). In northern Montana, ground nests were found in grass-dominated, rolling (greater than 10% slope) rangeland (Black 1992). Ground nests in South Dakota were always located in prairie in high condition (health) that was unused or lightly grazed (Lokemoen and Duebbert 1976). Ensign (1983) noted that ground nests were located in areas with sufficient vegetation to provide adequate cover from predators while at the same time

permitting adequate visibility. Ensign (1983) found that nests were selected near shrub cover that offered chicks and incubating birds additional concealment from predators.

Ferruginous hawks typically avoid nesting in dense tree stands (Dechant *et al.* 2002a). Lokemoen and Duebbert (1976) noted that 48% of 27 nests were located in trees, primarily tall cottonwoods (*Populus* spp.), that averaged 10.4 ± 2.6 m (range: 7.3 m to 14.6 m) above the ground. Lone trees or small, open groves were preferred for nest placement and all tree nests were situated within 1 km of prairie or grasslands (Lokemoen and Duebbert 1976). Similar findings were reported by Gilmer and Stewart (1983). Unlike ground nests in their study, Lokemoen and Duebbert (1976) noted that tree nests were not placed further from human activity than random, and land-use surrounding tree nests was not different from randomly selected units.

2.3.4 Foraging Habitat

Richardson's ground squirrels are the primary prey species for ferruginous hawks in Alberta. Schmutz (1989, 2008) showed that ferruginous hawk and ground squirrel populations peaked with small amounts of cultivation, but declined as cultivation exceeded 50% (Schmutz 1989). Therefore, productive foraging habitat for ferruginous hawks consists primarily of native prairie that supports high densities of Richardson's ground squirrels. This typically includes prairie with low cover values, moderately coarse textured soils and slopes of less than 10% (Downey 2003a, Downey 2004a).

2.3.5 Area Requirements

The average ferruginous hawk territory size is approximately 2.6 km^2 to 7.7 km^2 with a diameter of 1.6 km to 4 km (Call 1978). In South Dakota, Lokemoen and Duebbert (1976) found that ferruginous hawk nests were rarely closer than 2.6 ± 1.0 km. Olendorff (1993) report a mean "nearest neighbour" distance of 3.4 km based on 22 study years and 11 study areas. In Alberta, Schmutz (1977) found that ferruginous hawk nests were rarely closer than 0.8 km to each other. However, ferruginous hawks and Swainson's hawks in Alberta have been known to nest within 0.3 km of each other (Schmutz *et al*. 1980). This indicates that while ferruginous hawks demonstrate intraspecific competition, they do not exhibit interspecific competition with Swainson's hawks to the same degree. The average home range diameter for ferruginous hawks is 3.2 km to 3.4 km (Jasikoff 1982). Foraging distances from the nest site may vary with the availability of prey (Ensign 1983). Howard and Wolfe (1976) reported that eight out of nine hunting forays along the Utah-Idaho border were within 0.8 km of the nest site.

3 SWAINSON'S HAWK

3.1 Background

The Swainson's hawk (*Buteo swainsoni)* has historically been known as the most abundant raptor in western North America (Hull *et al.* 2008). This hawk species has extremely variable plumage, and similar to the ferruginous hawk, has both light- and dark-coloured phases (Semenchuk 1992). Distinguishing marks for Swainson's hawks include a dark band across the

chest and white areas on the sides of the rump in flight (Semenchuk 1992). In Canada, they breed from southern Yukon, through to western British Columbia, central and southern Alberta, Saskatchewan and southwestern Manitoba (Dechant *et al.* 2002b). Their breeding range extends through the western and central United States and into the Mexican states of Sonora and Durango (Dechant *et al.* 2002b).

Swainson's hawks were recently down-listed from 'Sensitive' to 'Secure' in Alberta under the General Status listing, and the species is not listed as 'Endangered' or 'Threatened' under the provincial *Wildlife Act* (GoA 2016). Nationally, Swainson's hawks have not been assessed by COSEWIC and are not included under Schedule 1 of *SARA* (COSEWIC 2016). Internationally, the Swainson's hawk is considered a species of 'Least Concern' by the IUCN (BirdLife International 2012b).

Although this species appears more adaptable to human disturbance than the ferruginous hawk, and while it seems to have adjusted to agricultural landscapes in many parts of its range, its numbers have declined in parts of the western United States and in the western Canadian prairie (England *et al*. 1997). Local population declines have been linked with decreases in its main prey species, Richardson's ground squirrel, the use of pesticides in parts of its winter range (England *et al.* 1997, GoA 2016) and nestling survival (Schmutz *et al.* 2006). Other limiting factors include habitat degradation due to intensive agriculture or urban development, shooting deaths, and collisions with vehicles (England *et al.* 1997). In addition, Wiggins *et al.* (2014) recently observed a negative relationship between Great Plains Swainson's hawk populations and grassland conversion to cropland not previously identified.

3.2 Ecology

Swainson's hawks migrate between breeding areas in North America and wintering grounds in South America (England *et al.* 1997). In Alberta, flocks begin to gather in preparation for the fall migration from their breeding grounds by late August and early September (England *et al.* 1997). Southerly migrations begin by mid-September (England *et al.* 1997). Hawks will generally return to Alberta and Saskatchewan between late April and early May. Swainson's hawks return to the breeding grounds after other raptor species such as red-tailed hawks (*Buteo jamaicensis*) and ferruginous hawks, both of which compete for similar nest sites (Andersen 1995). Swainson's hawks are typically monogamous, and pair bonds lasting for 10 years have been reported (England *et al.* 1997). In Alberta, Schmutz (1991) reported that Swainson's hawks did not change mates or territories in years following unsuccessful reproduction. Initiation of pair-bonding begins once birds return to breeding grounds. Nest building begins within seven to 15 days of arrival and typically lasts for approximately one week (England *et al.* 1997). Both members of a pair will help to build or refurbish a nest. Swainson's hawks will re-use nests built in previous seasons (Dechant *et al.* 2002b). Clutches are initiated from late May through to mid-June. Clutch size varies from one to four eggs, but typically consists of three or four (England *et al.* 1997). The incubation period lasts for five weeks between mid-May and early July (England *et al.* 1997). The female is primarily responsible for incubation, while the male delivers food to the nest. Fledging occurs approximately six weeks after hatching between late July and mid-August. Swainson's hawks are known to re-nest following an unsuccessful nesting attempt (Olendorff 1973).

Swainson's hawks are known to hybridize with other hawk species. Using mitochondrial DNA markers, Clark and Witt (2006) observed the first recorded occurrence of a Swainson's hawk and a rough-legged hawk (*Buteo lagopus*) hybrid. Similarly, Hull *et al.* (2007) observed hybrids (n = 3) between Swainson's hawks and red-tailed hawks. Two of the individuals in the latter study were identified in Alberta as a Swainson's hawk, whereas the third individual was identified in Utah as a red-tailed hawk. In both hybrid studies, overlapping breeding ranges are believed to have led to the interbreeding between species. It is not known if the offspring of Swainson's hawks and red-tailed hawks are fertile (Hull *et al.* 2007).

3.2.1 Diet

As with ferruginous hawks, the Richardson's ground squirrel is the main prey species of Swainson's hawks throughout Alberta and Saskatchewan (Houston and Schmutz 1995). Ground squirrels averaged 69% of the total prey items for Swainson's hawks near Hanna, Alberta (Schmutz *et al.* 1980). Like ferruginous hawks, declines in Swainson's hawk density and productivity have been correlated with declines in abundance of ground squirrels in southeastern Alberta and southwestern Saskatchewan (Schmutz and Hungle 1989, Houston and Schmutz 1995). Unlike ferruginous hawks, however, Swainson's hawks typically prey primarily on juvenile and not adult ground squirrels, and as a result have a broader diet early in the season (Schmutz and Hungle 1989, Dechant *et al.* 2002b). Swainson's hawks rely on other prey, in particular voles (*Microtus* spp.), early in the season during the egg formation and egg-laying periods when juvenile squirrels are unavailable (Schmutz and Hungle 1989). As some vole species have been found in high numbers in cultivated fields, this may explain why Swainson's hawks, unlike ferruginous hawks, utilize areas with higher amounts of cultivation (Schmutz and Hungle 1989).

Northern pocket gophers, meadow voles, cottontails (*Sylvilagus* spp.) and juvenile lagomorphs (rabbits and hares) as well as ground-nesting birds are among the other common prey species consumed by Swainson's hawks across its breeding range (Gilmer and Stewart 1984, Andersen 1995, England *et al.* 1997, Gerstell and Bednarz 1999). Swainson's hawks are also known to forage opportunistically on grasshoppers (Johnson *et al.* 1987).

3.2.2 Predators

Current literature is limited with respect to Swainson's hawk predators; however, owls are suspected to prey on eggs and nestlings. Gilmer and Stewart (1984) attributed 19% of nest failure to predation; however, no predator species were identified in their study. Cannibalism is suspected to occur in the Swainson's hawk; however, no accounts have been documented (Pilz 1976).

3.3 Habitat Requirements

3.3.1 General

Swainson's hawks typically occupy open grasslands with scattered trees or clumps of trees and shrubs (Dechant *et al.* 2002b). Swainson's hawks are fairly flexible in their habitat use and use native shortgrass, mixedgrass, fescue, tallgrass, and sandhill prairies as well as pastures, hayland,

and cropland (Dechant *et al.* 2002b). Within grassland habitats, Swainson's hawks will make use of riparian areas, isolated trees, and shelterbelts (Dechant *et al.* 2002b). Swainson's hawks are also found in the aspen parklands of the Canadian prairies (Dechant *et al.* 2002b).

3.3.2 Breeding Habitat and Land-use

In comparison to ferruginous hawks, Swainson's hawks breed in areas with a greater percentage of cultivation and are apparently more tolerant of human activity. An interspersion of cultivated land with native grasslands appears to have allowed Swainson's hawks to take advantage of easier prey accessibility in harvested fields (Bechard 1982). Other human changes to the landscape, such as planted shelterbelts or trees around abandoned farmsteads, also appear to have provided Swainson's hawks with greater nesting opportunities than previously available (Olendorff 1973, Gilmer and Stewart 1984, Groskorth 1995).

Schmutz (1984) examined habitat use in 41 km^2 plots in southeastern Alberta in which cattle grazing or cultivation of cereal crops was the dominant land-use. Unlike ferruginous hawks, Swainson's hawk nest density was shown to be higher in areas with 11% to 30% cultivation than in areas with less cultivation. Schmutz (1984) also found that Swainson's hawks nested in closer proximity to human habitation than ferruginous hawks. In comparison, in south-central North Dakota, Gilmer and Stewart (1984) report that mixedgrass prairie and hayland were the dominant (75%) land-uses within both 100 m and 1.0 km of a sample of 27 nests. Gilmer and Stewart (1984) noted that within their study area some Swainson's hawk pairs were found to nest successfully in sites with intensive agriculture and human activity. In an area with little remnant natural grassland, the Regina Plain in Saskatchewan, Groskorth (1995) found that Swainson's hawks selected nest sites with more surrounding grassland, trees and shrubs and significantly fewer wheat fields within 1 km of the nest site than random sites. Smallwood (1995) reported that Swainson's hawks in the Sacramento Valley, California "preferred" riparian habitat, grassland, alfalfa stands more than 2 years old during irrigation and mowing, and annual field crops during harvest. Hawks "avoided" tilled fields, irrigated pasture, annual field crops and developed areas.

3.3.3 Nesting Habitat

Swainson's hawks most often nest in trees and shrubs that occur in isolation, are clumped or form parts of shelterbelts (Dechant *et al.* 2002b, Inselman *et al.* 2015). Trees used for nesting range in height from 2 m to 22 m (Dechant *et al.* 2002b). Swainson's hawks will use artificial nest platforms and occasionally nest on man-made structures such as telephone poles; however, not as commonly as ferruginous hawks (Gilmer and Stewart 1984, England *et al.* 1997, Dechant *et al.* 2002b). Swainson's hawks have been reported to nest on the ground in areas where no suitable trees or structures are available (Dechant *et al.* 2002b).

In North Dakota, Gilmer and Stewart (1984) reported that the most common nesting sites were located in shelterbelts (43%) and that cottonwoods (*Populus deltoides*) were the most commonly used (43%) trees for nesting. In total, nests in shelterbelts and in trees adjacent to wetlands accounted for approximately 65% of all nest sites. Gilmer and Stewart (1984) also noted that approximately 75% of all nests sites were directly or indirectly produced by humans.

3.3.4 Foraging Habitat

Swainson's hawks use native prairie, cropland and haylands for foraging (Dechant *et al.* 2002b). Use of these habitats varies through the season in relation to ease of prey detection and availability of juvenile ground squirrels. Recently harvested cropland and mowed hayland provide favoured foraging habitat at the beginning and end of the season when hunting success in these areas is improved due to ease of prey detection (Bechard 1982, Schmutz 1987).

3.3.5 Area Requirements

Estimates of Swainson's hawk home range size vary from 6.2 km^2 to 27.3 km^2 (Dechant *et al.* 2002b). In Alberta, Schmutz (1977) reported a minimum radius of 0.35 km for nesting territories. This calculation was based on the assumption that nesting territories were circular and nest sites were in the center of the territory. In Colorado, the reported home ranges for male hawks (*i.e.*, 31.7 km^2) were greater than those recorded for females (*i.e.*, 19.9 km^2) (Andersen 1995).

4 GOLDEN EAGLE

4.1 Background

The golden eagle is the largest of the raptor species considered in this report. It has a wingspan of up to 2.2 m and can weigh as much as 6.1 kg (Kochert *et al.* 2002). Adult golden eagles are entirely dark brown except for the nape and hind neck, which are golden brown (Semenchuk 1992). Historically, golden eagles inhabited much of North America (Bechard and McGrady 2002). Today, this species has essentially been eliminated from most eastern states; however, nesting populations in Alaska, Canada, and the majority of the western United States are considered to be stable (Bechard and McGrady 2002, Kochert and Steenhof 2002). Golden eagles breed mainly along the lower reaches of the major river systems in southern Alberta and in the Rocky Mountain region of the province (Semenchuk 1992). Golden eagles are uncommon in northern Alberta (Semenchuk 1992).

In Alberta, golden eagles are considered a 'Sensitive' species under the General Status listing (GoA 2016). The species is considered 'Not at Risk' federally, a status that has remained unchanged since 1996 (COSEWIC 2016). The species is not listed as 'Endangered' or 'Threatened' under the *Wildlife Act* or *SARA*. Internationally, the golden eagle is listed as a species of 'Least Concern' by the IUCN (BirdLife International 2015a).

There are an estimated 100 to 250 breeding pairs of golden eagles in Alberta (GoA 2016). Because of its low population numbers and dispersal over a large area of the province, golden eagles are considered at heightened risk of extirpation (GoA 2016). A recurring concern is the effect of low population density of the species over large areas, which may also be contributing to the decline of this species (Alberta Environment and Parks [AEP] 2016).

In Alberta, disturbance from human-related activities is considered the greatest threat to golden eagles. Elsewhere, declines in some nesting populations of eagles in the intermountain western United States, such as in Utah and in the Snake River region of Idaho, have been directly

associated with loss of sagebrush steppe and diminishing jackrabbit (*Lepus* spp.) populations (Gindrod 2001, Kochert and Steenhof 2002). In general, habitat destruction due to urbanization, industrial and intensive agricultural development as well as several other forms of direct and indirect human disturbance are the primary factors responsible for declines in this species. Kochert and Steenhof (2002) found that 73% of eagle deaths from the early 1960's to the mid-1990's were human-related, including: accidental trauma (27%), electrocution (25%), shooting (15%) and poisoning (6%). Accidental trauma encompasses deaths due to collisions with vehicles, fences, wires and wind turbines. Turbine blade strikes are attributable to the deaths of between 28 to 43 eagles each year in the Altamont Pass Wind Resource Area in west-central California (Kochert and Steenhof 2002). Harness (1997) reported 272 eagle electrocutions in the western U.S. and Canada between 1986 and 1996. Electrocution is a greater threat to juvenile eagles (Snow 1973). Lead is the most common cause of poisoning deaths of golden eagles (Wayland and Bollinger 1999). Lead shot or bullets may be one of the main sources of lead poisoning (Wayland and Bollinger 1999, Kochert and Steenhof 2002). Agricultural pesticides, primarily organo-phosphates and carbamates, account for the majority of the remaining poisoning deaths. These pesticides are often ingested through consumption of other animals poisoned with these chemicals. Another form of human induced mortality is nest abandonment due to direct human disturbance of nests during incubation (Boeker and Ray 1971, Snow 1973).

Recently, some researchers have refuted the reported declining population trends of golden eagles in western North America. For example, Millsap *et al.* (2013) observed a 0.4% increase in the golden eagle population in the western United States from 1968 to 2010 and an increase of 0.5% from 1990 to 2010. In a commentary by McCaffery and McIntyre (2005), it was suggested that data collected by Hoffman and Smith (2003) do not show declining trends in western North America. Watson (2010) noted that the North American population trends are largely unknown and more data is required to properly evaluate the species. In addition, Watson (2010) suspects that the population is closely tied with 10 year fluctuations of jackrabbit populations. Sufficient data is not yet available to capture these cycles and distinguish them from the overall population trend (Watson 2010). Furthermore, the 2007 edition of *The Atlas of Breeding Birds of Alberta* identified the golden eagle as having an unchanged distribution and increasing relative abundance (Federation of Alberta Naturalists 2007).

4.2 Ecology

Eagles return to breeding areas from late March to mid-May (Kochert *et al.* 2002). Golden eagles appear to demonstrate fidelity to breeding territories regardless of nesting success (Snow 1973). Nest defense by golden eagles is variable (Snow 1973). Tolerance toward immature eagles by breeding pairs has been shown; however, other adult eagles are not typically tolerated in nesting areas. Clutch size usually ranges from one to three eggs (Snow 1973). The female eagle does most of the incubating; however, males will assist with incubation on occasion (Snow 1973). Incubation is thought to be the most critical period during which eagles will desert the nest (Snow 1973). The chance of nest desertion decreases after the young have hatched. The incubation period ranges from 35 to 43 days (Snow 1973). Reported hatching success rates for golden eagles in Montana, Colorado and Idaho, ranged from 1.13 young/nest to 2.1 young/nest (Snow 1973). For the first few weeks after hatching, at least one parent is present at the nest at all times, typically the female (Snow 1973). Fratricide is fairly common among golden eagle

chicks during the first three weeks due to the size difference of eaglets that are hatched two to four days apart (Snow 1973). Chicks will fledge at nine to 10 weeks; however, they will remain dependent on their parents until they are at least 100 days old (Snow 1973). Golden eagles leave northern areas from September to early October to begin the southerly migration to their wintering grounds (Kochert *et al.* 2002).

4.2.1 Diet

In general, golden eagles show a strong preference for small mammal prey, particularly rabbits and rodents (Snow 1973). Jackrabbits and cottontails are commonly reported as the main prey items for golden eagles (McGahan 1968, Boeker and Ray 1971, Snow 1973, Marzluff *et al.* 1997a). In Montana, for example, McGahan (1968) found that whitetail jackrabbits (*Lepus townsendii*), desert cottontails (*Sylvilagus audubonii*) and mountain cottontails (*Sylvilagus nuttallii*) comprised 69.8% of the total prey taken by eagles. Similarly, Boeker and Ray (1971) found that jackrabbits and cottontail rabbits comprised more than 75% of the golden eagles total diet. Bates and Moretti (1994) found that the number of young produced by golden eagles from 1982 to 1992 in eastern Utah was correlated with rabbit abundance. Although alternative smaller prey species such as Richardson's ground squirrels and grouse are more commonly taken, the golden eagle is a powerful hunter and predation on livestock has occasionally been reported (Dekker 1985, Phillips *et al.* 1996). Golden eagles have been known to prey on young lambs and goats as well as killing larger prey such as adult mule deer (*Odocoileus hemionus*), pronghorn (*Antilocapra Americana*), coyotes (*Canis latrans*), domestic calves and sheep (Woodgerd 1952, McGahan 1968, Phillips *et al.* 1996; Hamel and Cote 2009). A case of severe golden eagle predation on domestic calves was reported in Socorro County in central New Mexico (Phillips *et al.* 1996). In this case, six calves were confirmed to have been killed and 48 injured by golden eagles from 1987 to 1989 (Phillips *et al.* 1996). However, typically the impact of golden eagles on ungulate populations is negligible (Snow 1973). Golden eagles will also feed on carrion, making them susceptible to pesticide or lead poisoning (Snow 1973). Brown and Watson (1964) suggest that food supply does not affect golden eagle density as eagles maintain a sufficiently large home range so that a critical food level is rarely reached. Although rare, accounts of kleptoparasitism by golden eagles have been reported in the literature (Jung *et al.* 2009).

4.2.2 Predators

Current literature does not specifically identify predators for golden eagles; however, crows and common ravens are suspected to prey on eggs.

4.3 Habitat Requirements

4.3.1 General

Golden eagles occur in open country, including shrublands and grasslands, and prefer elevated nest sites, usually cliff ledges, near hunting areas (Grindrod 2001). In Alberta, golden eagles nest on cliffs along prairie rivers or on rocky ledges in the Rocky Mountain region (Semenchuk 1992). Golden eagles occupy definite breeding territories that include feeding, roosting, nesting and "soaring-playing" areas (Snow 1973).

4.3.2 Breeding Habitat and Land-use

Marzluff *et al.* (1997a) measured spatial use and habitat selection of radio-tagged golden eagles at eight to nine territories from 1992 to 1994 in the Snake River Birds of Prey National Conservation Area in Southwestern Idaho. Most golden eagle home ranges contained more big sagebrush (*Artemisia tridentata*) or yellow rabbitbrush (*Chrysothamnus viscidiflorus*), more cliff or rock outcrop and less grassland and agriculture than expected based on availability. Within their home ranges, golden eagles concentrated their activity within smaller core areas. Avoidance of agricultural lands was significant within 90% of core areas and was consistent among individuals during the breeding and non-breeding seasons.

4.3.3 Nesting Habitat

The large stick nests constructed by golden eagles are usually located on cliff ledges or shelves or rocky bluffs (Snow 1973, Semenchuk 1992). Golden eagles will also nest in trees and occasionally on the ground. Boeker and Ray (1971) conducted a golden eagle nesting study along the Front Ranges of the Rocky Mountains in New Mexico, Colorado and Wyoming. They found that 93% of the 150 nests located during this study were situated on cliffs and the remainder were in trees or on earthen mounds. Cliffs overlooking open grasslands were more heavily used than those overlooking closed forests. Similarly, in Montana, McGahan (1968) found that 62% of nests observed were located on cliffs. Douglas-fir (*Pseudotsuga menziesii*) trees with large enough limbs to support heavy, bulky nests provided the next most common nest site (McGahan 1968).

Height of golden eagle tree nests can vary from 3 m to 30 m above the ground (Snow 1973). Golden eagles will often have at least two to three alternate nest sites (Snow 1973). Pairs may use the same nest during consecutive nesting seasons; however, they often repair and use alternate nests (Snow 1973). Aspect may play a role in nest site selection. McGahan (1968) proposed that south- and east-facing sites are superior as maximum sun exposure is especially important during the early spring incubation period when temperatures can drop below freezing. South- and east-facing sites receive morning sun, with easterly exposures having the added advantage of afternoon shade.

4.3.4 Foraging Habitat

Golden eagles typically forage in open habitats, including grasslands and steppe-like vegetation (Kochert *et al.* 2002). Marzluff *et al.* (1997a) reported that golden eagles in southwestern Idaho selected shrub habitats during foraging and avoided disturbed areas, grasslands and agriculture. Shrub habitat had the greatest potential to contain their principal prey, black-tailed jackrabbits. Selection for areas with abundant and large shrub patches as key foraging areas was particularly apparent for golden eagles in highly fragmented or dispersed shrublands.

4.3.5 Area Requirements

Golden eagle territory size is dependent to some extent on the availability of food, nest sites and suitable terrain for flying (Snow 1973). Marzluff *et al.* (1997a) reported that golden eagle home ranges in Idaho ranged from 1.9 km^2 to 83.3 km^2 during the breeding season and from 13.7 km^2 to 1,700 km^2 outside of this season. However, Marzluff *et al.* (1997a) noted that activity was

concentrated in small core areas of 0.3 km^2 to 15.35 km^2 and 4.85 km^2 to 63.8 km^2 during the breeding and non-breeding seasons, respectively. Dixon (1937) mapped the territories of 27 pairs of golden eagles in California; territories in that state ranged from 49.2 km^2 to 152.8 km^2, with an average of 93.2 km^2. Dixon (1937) found that territories in hilly terrain were smaller than territories established in flat, open country. Reynolds (1969) studied a golden eagle pair in south-central Montana and reported that the pair spent most of their time in a 33.7 km^2 area, but used a total area of 82.9 km^2. McGahan (1968) reported that the distance between neighboring eagles in Montana ranged from 1.6 km to 16.9 km.

5 SHORT-EARED OWL

5.1 Background

The short-eared owl is a medium-sized owl with streaked upper parts, short ear tufts, small facial disc and distinctive black patches on the underwings near the wrists (Semenchuk 1992). The short-eared owl is the most widely distributed owl in the world (Federation of Alberta Naturalists 2007), breeding throughout much of the northern half of North America and migrating as far south as Mexico and the West Indies (Clayton 2000, Dechant *et al.* 2002c). In Canada, short-eared owls breed in every province and territory from the southern border to the low Arctic; however, they are absent from the Boreal Forest and other heavily forested areas (Cadman and Page 1994). Short-eared owls are nomadic and respond irruptively to high concentrations of small mammals (Clark 1975, Holt and Leasure 1993, Clayton 2000, Dechant *et al.* 2002c).

The short-eared owl is currently listed as ‘May Be At Risk’ in Alberta under the General Status listing (GoA 2016). The species has not been listed under the provincial *Wildlife Act*. Nationally, the short-eared owl was assessed by COSEWIC in the mid-1990s and given the status of ‘Special Concern’ (Cadman and Page 1994). This status was re-confirmed in 2008 and the species remains a species of ‘Special Concern’ in Canada (COSEWIC 2008b, 2016). *Asio flammeus* is listed as a species of ‘Least Concern’ by the IUCN (BirdLife International 2015b).

There are no current, accurate short-eared owl population estimates in Alberta; however, populations of this species are thought to be declining in Alberta (GoA 2016). Breeding Bird Survey (BBS) data show a long-term, although non-significant, decline of this owl species in Alberta, and significant declines in short-eared owl abundance are evident from BBS routes across Canada (Clayton 2000). The most severe short-eared owl population declines in North America have occurred in the northeastern United States (Clayton 2000). Short-eared owl population assessments and breeding habitat evaluations in Canada are complicated by the lack of information from remote northern areas and the owl’s irruptive population trends (Cadman and Page 1994). Around the turn of the century, in Alberta the short-eared owl was described as “common” along the Milk River and the West Butte (Macoun and Macoun 1909, cited in Cadman and Page 1994).

The main reason for short-eared owl declines in North America is generally agreed to be the loss and degradation of habitat due to agricultural, industrial, recreational and urban development (Cadman and Page 1994, Clayton 2000, Dechant *et al.* 2002c, COSEWIC 2008b). Heavy grazing

over large areas is also cited as a factor contributing to habitat degradation (Clayton 2000). Its ground-nesting habit and nomadism makes this species particularly vulnerable to habitat loss (Holt and Leasure 1993). In general, food abundance is clearly linked with short-eared owl population fluctuations (Holt and Leasure 1993, Clayton 2000, Dechant *et al.* 2002c). Meadow voles are considered the main "predictive resource" in the Canadian prairies (Clayton 2000). Pesticide use may also be a limiting factor to short-eared owls, but is a topic that requires further study (Holt and Leasure 1993, Cadman and Page 1994, Clayton 2000). Other limiting factors due to human activity include: shootings; collisions with vehicles, radio antennas or high-tension guy wires; entanglement in barbed wire; and nest destruction by farm machinery (Clark 1975, Cadman and Page 1994).

5.2 Ecology

Short-eared owls are solitary or communal during the non-breeding season, but are generally considered loosely colonial breeders (Holt and Leasure 1993). A nomadic species, short-eared owls exhibit seasonal monogamy (Holt and Leasure 1993, COSEWIC 2008b). Pair-bonding begins in late winter (*i.e.*, mid-February) as communal roosts disband (Holt and Leasure 1993). Males perform an elaborate sky dance display for prospective females. Short-eared owl migrants return to Alberta during March and early April; however, these owls are also known to overwinter in Alberta as far north as Edmonton and Grande Prairie (Clayton 2000). In arctic areas, coastal marshes and interior grasslands across North America (COSEWIC 2008b), these owls usually breed from early April to late August (Dechant *et al.* 2002c). Nesting may begin as early as late March in areas where wintering and breeding grounds overlap, with nests built by the female (Dechant *et al.* 2002c, Jehl 2004). Egg-laying can begin as early as April and last through early June (COSEWIC 2008c). Generally correlated with resources abundance (Clark 1975), clutch sizes in North America range from three to 11, with a reported mean clutch size of seven (Wiggins 2004). Eggs are usually laid at one to two day intervals and are incubated for 24 to 29 days (Holt and Leasure 1993, Clayton 2000). In North Dakota, hatching dates ranged from early May to late July, with a mean hatch date of mid-June (Dechant *et al.* 2002c). Incubation and brooding are performed by the female, while males are responsible for providing food. Short-eared owls may re-nest if the first clutch is destroyed (Dechant *et al.* 2002c). Owl chicks usually leave the nest when they are between 14 to 17 days old and wander up to 200 m away (Holt and Leasure 1993, Clayton 2000). Fledging occurs when owls reach 27 to 35 days of age (Holt and Leasure 1993). Wiggins (2004) reported 50% success rate for fledging short-eared owls. The short-eared owl is known for its ability to breed sooner and increase its clutch size in times of prey abundance (Clark 1975, Holt and Leasure 1993, Cadman and Page 1994, Dechant *et* al 2002c).

5.2.1 Diet

Short-eared owls forage primarily at dusk and dawn (COSEWIC 2008b). Small mammals, in particular *Microtus* voles, dominate the short-eared owl's diet throughout its North American range (Holt and Leasure 1993, Clayton 2000). The short-eared owl's diet does not vary much by season, sex or age of individuals (Holt and Leasure 1993). A strong correlation between vole abundance and owl abundance has been demonstrated (Holt and Leasure 1993). Meadow voles (*Microtus pennsylvanicus*) were the predominant prey in seven of nine studies from Canada and

the United States, constituting 78% to 97% of prey items (Holt 1993). Meadow vole populations undergo a characteristic two- to five-year cyclic fluctuation in density. The mechanisms responsible for causing vole peaks over large areas and the geographic extent of these population booms are not well understood (Clayton 2000). In other North American studies where meadow voles were absent or low in abundance, important prey species included: deer mice (*Peromyscus maniculatus*); California voles (*Microtus californicus*); Hispid cotton rats (*Sigmodon hispidus*); and least shrews (*Cryptotis parva*) (Clayton 2000). Holt (1993) also noted that pocket gophers (*Thomomys* spp.), mice (*Peromyscus* spp.), kangaroo rats (*Dipodomys* spp.) and lemmings occasionally make up part of the short-eared owl's diet.

The fluctuation of short-eared owl population numbers in response to peaks or ebbs in vole population cycles is well-documented (Clayton 2000). Clayton (2000) cites an example of this phenomenon in southeastern Alberta, where an apparent increase in the occurrence of owls was correlated with an increase in the abundance of voles in the area as determined by small mammal trapping. In Saskatchewan, Poulin *et al.* (1998) documented a similar synchronous increase in the short-eared owl population in response to a dramatic one-year increase in the vole population. Short-eared owls hunt primarily at night; however, diurnal hunting has also been reported (Clayton 2000). Hunting is usually done "on the wing", flying less than 3 m above the ground (Holt and Leasure 1993). Hovering at higher altitudes (up to 30 m) has also been reported, and less frequently, owls have been noted to use perches for hunting (Holt and Leasure 1993).

5.2.2 Predators

As short-eared owls are ground-nesters and use open habitats, they are primarily vulnerable to mammalian predation by species such as red foxes (*Vulpes vulpes*), striped skunks (*Mephitis mephitis*) (Holt and Leasure 1993), feral cats (*Felis catus*) and dogs (*Canis familiaris*), great horned owls (*Bubo virginianus*), snowy owls (*Nyctea scandiaca*), red-tailed hawks, rough-legged hawks, northern harriers (*Circus cyaneus*), norther goshawks (*Accipiter gentilis*), peregrine falcons (*Falco peregrinus*), herring gulls (*Larus argentatus*) and common ravens (*Corvus corax*) (COSEWIC 2008b).

5.3 Habitat Requirements

5.3.1 General

In general, the short-eared owl occurs in extensive areas of open, unforested habitat, including: intact grasslands, arctic tundra, sand-sage, fallow pastures and occasionally fields planted with row-crops (Cadman and Page 1994, Clayton 2000, Dechant *et al.* 2002c, Wiggins *et al.* 2006, COSEWIC 2008b). The species' main breeding requirement is for sufficient prey adjacent to suitable nesting habitat. Short-eared owls breed in open habitats such as native prairie, haylands, small-grain stubble, wet-meadows, marshlands, peatlands and clear-cuts throughout the non-mountainous regions of Alberta (Clayton 2000, Dechant *et al.* 2002c). The majority of short-eared owl reports are from the Grassland and Parkland natural regions of Alberta (Semenchuk 1992, Clayton 2000). Short-eared owls will generally sleep, nest and roost on the ground (Holt and Leasure 1993). In winter owls will tree-roost (Holt and Leasure 1993). Long grass is also

often used for roosting (Cadman and Page 1994). Limited specific information is available about historical breeding sites and habitat associations of short-eared owls in Alberta (Clayton 2000).

5.3.2 Nesting Habitat

Unlike the majority of North American owls, the short-eared owl generally nests on the ground (Holt and Leasure 1993, Clayton 2000, Dechant *et al.* 2002c). Nests are usually located on open, dry, upland sites; however, wetter lowlands, such as peat bogs and wetlands are used occasionally (Dechant *et al.* 2002c, COSEWIC 2008b). The species composition and structure (height and density) of the vegetation in which short-eared owl nests are found varies substantially across the owl's range (Dechant *et al.* 2002c). Not surprisingly, the degree of concealment of nests varies accordingly. The apparent diversity of preferred nesting habitat may be an indication of the importance of prey availability in the distribution of breeding sites annually (Clayton 2000).

In general, grass cover is preferred for nesting. Based on an assessment of 63 short-eared owl nests in North America, Clark (1975) found that the majority of nests were located in grassland (55%), followed by grain stubble (24%). Of the remainder of nests, 14% were found in hayland and 6% in shrubs (*i.e.*, buckbrush [S*ymphoricarpos occidentalis*]). Jehl (2004) found that dense grassland was preferred for nesting, as well as tundra with small willows; however, nest choice was largely dependent on small mammal prey. Based on the results of a 50-year banding program in south-central Saskatchewan, Houston (1997) reported that short-eared owl nests were concentrated in open stubble habitats. In Montana, of the vegetation within 15 m of 28 nests, 85% was grasses, 8% was herbs and 7% was a combination of herb and grass. Ninety percent of this vegetation was less than 0.5 m tall, 9% was between 0.5 m to 1.0 m tall and 1.0% was greater than 1 m tall (Holt and Leasure 1993). In two intensively managed grassland complexes in southeastern Illinois, short-eared owls preferred to nest in grass heights of 30 cm to 40 cm, such as in rotary-mowed fields (Herkert *et al.* 1999). In South and North Dakota, the majority of short-eared owls nests were found well-concealed in undisturbed grass-legume vegetation 30 cm to 60 cm tall (Duebbert and Lokemoen 1977). Nesting in areas dominated by buckbrush and herbaceous vegetation has also been reported in northwestern North Dakota (Murphy 1993, cited in Dechant *et al.* 2002c).

5.3.3 Foraging Habitat

Short-eared owls forage in open areas that support cyclic small mammal populations, in particular voles (Clayton 2000). Few studies, however, have specifically evaluated foraging habitat use by this species. Short-eared owls in an agricultural landscape in southern Chile were found to concentrate their hunting along roadsides, in ungrazed meadows and in untilled lands (Martinez *et al*. 1998). Several studies have found that the predominant short-eared owl prey, meadow voles, prefer native prairie or undisturbed meadows with greater amounts of vegetative cover and typically avoid cultivated fields (Marinelli and Neal 1995, Peles and Barrett 1996, Basquill and Bondrup 1999, Getz *et al.* 2001, Lin and Batzli 2001). Additional research is needed to determine meadow vole densities in stubble habitats, such as in south-central Saskatchewan where short-eared owl banding programs have been conducted (Houston 1997).

5.3.4 Area Requirements

Short-eared owls are generally associated with large, open expanses of grassland (Holt and Leasure 1993, Dechant *et al.* 2002c). Herkert *et al.* (1999) suggest that short-eared owls may respond to the total amount of grassland within the landscape rather than the size of individual grassland tracts. Owls may therefore use small blocks of habitat if the blocks are located near to other more extensive areas of grassland. There is a wide range in the size of documented short-eared owl breeding territories (Holt and Leasure 1993). In Manitoba, breeding territories averaged 0.82 km^2 and ranged from 0.23 km^2 to 1.21 km^2 (Clark 1975, Holt and Leasure 1993). Clark (1975) suggested that breeding territory size may be inversely related to vole density, (*i.e.*, breeding territory size increases with decreasing prey (vole) densities and decreases with increasing prey densities).

6 PRAIRIE FALCON

6.1 Background

The prairie falcon is similar in size the peregrine falcon but is paler in colour (Semenchuk 1992). Prairie falcons have the characteristic black malar streaks beneath the eyes as well as dark ear-patches and distinctive white areas between the eyes and ear-patches (Steenhoff 2013). This species is distinguished in flight from below by its distinctive dark axillaries and trailing edge of underwing-coverts, which contrast with light-colored underwing surface (Steenhoff 2013). Prairie falcons breed in suitable cliff habitats along rivers or waterbodies in the Grassland, Parkland and occasionally the Rocky Mountain Natural Region of southern Alberta (Semenchuk 1992, Paton 2002, NRC 2006). Their breeding range in Alberta extends as far north as Red Deer, with nesting areas primarily in cliffs and hoodoos found along the Bow, Red Deer, Milk, South Saskatchewan and Oldman Rivers and their tributaries (Paton 2002). High densities of prairie falcons and prairie falcon nests occur in sandstone cliffs and hoodoos along the Milk River from the town of Milk River to Writing-on-Stone Provincial Park and in Police Coulee (Quinlan *et al.* 2003, Downey and Quinlan 2004). Prairie falcons occur to a lesser extent in the dry south-central interior of British Columbia and in southwestern Saskatchewan. Prairie falcons are found year-round in the western and central United States, and their wintering range extends into Mexico.

The prairie falcon is ranked as ‘Sensitive’ in Alberta under the General Status listing (GoA 2016). Under the provincial *Wildlife Act*, the species is considered a species of ‘Special Concern’. Nationally, the prairie falcon is considered ‘Not at Risk’ by COSEWIC and the species is not included under *SARA* (Kirk and Banasch 1996, COSEWIC 2016). Internationally, the prairie falcon is considered a species of ‘Least Concern’ by the IUCN (BirdLife International 2012c).

The prairie falcon population in Canada was estimated at 250 to 500 pairs in the mid-1990s (Kirk and Banasch 1996). The population at the time was thought to be stable, with possible local declines in Alberta (Kirk and Banasch 1996). The most recent provincial estimate was completed in 2002, at which time there was estimated to be 200 to 250 breeding pairs in Alberta (Paton 2002). The minimum estimated population of prairie falcons in Alberta in 2002 was 202 pairs,

which represents approximately 81% of the Canadian population and 6% of the continental population (Paton 2002).

Prairie falcon surveys in Alberta conducted in the early 2000s showed a reduction of successful breeding pairs along stretches of the Bow River near Calgary at the Bassano Dam (Paton 2002). There has also been a reported reduction in the number of breeding pairs in the Milk River Natural Area (Paton 2002). A decreasing number of prairie falcon nests were observed along the Milk River during recent aerial surveys for raptors conducted in 2000, 2002 and 2003 (Erickson 2000, Quinlan *et al.* 2003, Downey and Quinlan 2004). Surveys conducted in 2000 were carried out as part of the provincial peregrine falcon survey (Erickson 2000). Surveys were conducted along the Milk River and North Milk River from the points where the two rivers enter Alberta to approximately 15 km below Writing-on-Stone Provincial Park (Quinlan *et al.* 2003, Downey and Quinlan 2004). Nineteen nests were found in this area in 2000, while 15 and 13 nests were found in 2002 and 2003, respectively (Erickson 2000, Quinlan *et al.* 2003, Downey and Quinlan 2004). In general, there is a need for improved survey protocols and more consistent monitoring of core populations of prairie falcons in Alberta to provide a reliable index of population and productivity trends (Paton 2002).

As prairie falcons in Alberta are at the northern limit of their continental range, their populations are subsequently vulnerable to environmental and habitat changes (Paton 2002). Secure nesting cliffs near to prairie grasslands with an adequate prey base of Richardson's ground squirrels are essential to maintaining prairie falcon populations (Paton 2002, GoA 2016). Habitat loss due to agricultural, urban, industrial and recreational development is the primary limiting factor for this species. Human induced or natural erosion and flooding is also responsible for loss of nesting sites. Organo-chlorine pesticide contamination and direct human disturbance from shooting or nest disturbance in the early stages of the breeding season are other commonly cited limiting factors (Snow 1974, Paton 2002).

A provincial Conservation Management Plan was released by AEP in 2012, covering the years 2012 to 2017 (Paton 2012). This plan outlines the goals and objectives of prairie falcon management in Alberta as well as ongoing research and population monitoring needs.

6.2 Ecology

In years with mild winters and adequate prey, adult prairie falcons will winter throughout much of their southern Alberta breeding range (Paton 2002). Juveniles, however, typically migrate to the United States and northern Mexico (Paton 2002). Courtship and mate selection occurs on the breeding grounds several weeks prior to egg-laying. In Alberta, breeding territory establishment begins in March and egg-laying occurs in mid- to late April (Paton 2002). Usually four to five eggs are laid; however, clutch sizes range from two to six eggs (Paton 2002). The eggs are incubated by the female for a period of 29 to 33 days. Prairie falcons will re-nest if the first clutch is destroyed early in the incubation. Prairie falcon young typically fledge at 36 to 41 days, and most are fledged by the third or fourth week of June (Paton 2002). The number of fledglings per nesting attempt is highly variable for falcons across their range, varying from 1.2 to 3.9 young per territorial pair (Paton 2002).

6.2.1 Diet

Richardson's ground squirrels are the dominant component of prairie falcon diets in Alberta (Paton 2002). A study along the Bow River reported that ground squirrels accounted for 89% of the biomass fed to prairie falcon young (Hunt 1993). Fluctuations in the number of successful prairie falcon pairs on the Oldman River Reservoir between 1990 and 1997 appeared to correspond with fluctuations in ground squirrel populations in the area (Fyfe 1997). Similar results have been reported in southwestern Idaho, where researchers have found that falcon reproduction is influenced by abundance of Townsend's ground squirrels (*Spermophilus townsendii*) (Steenhof and Kochert 1988, Marzluff *et al.* 1997b, Steenhof *et al.* 1999). Prairie falcons, across their range, also consume lesser amounts of other small mammals, small- to medium-sized birds, reptiles and insects (Paton 2002). Based on research from a section of the Bow River, ground squirrels made up 68% of the prairie falcon's diet; grassland birds such as horned larks (*Eremophila alpestris*), western meadowlarks (*Sturnella neglecta*) and European starlings (*Sturnus vulgaris*) made up 27%; and small mammals made up 5% (Paton 2002). Consumption of alternate prey items has been shown to increase in years of low ground squirrel numbers (McFadzen and Marzluff 1996, Paton 2002). However, studies suggest that in years of lower ground squirrel abundance, prairie falcon productivity was found to be lower (Marzluff *et al.* 1997b, Steenhof *et al.* 1999, Paton 2002). It is not known how many ground squirrels are needed to support a productive population of prairie falcons (Paton 2002).

6.2.2 Predator

In southwestern Idaho, McFadzen and Marzluff (1996) identified predation as a major cause of mortality and named great horned owls and golden eagles as the two predators responsible in a two-year study of fledgling success. Ectoparastisim was also determined to be an influencing factor in success (McFadzen and Marzluff 1996). Ogden and Horocker (1977) attributed 23% of egg losses to predation, identifying coyotes and bobcats (*Lynx rufus*) as know predators to the species.

6.3 Habitat Requirements

6.3.1 General

Prairie falcons occur in southern Alberta primarily in areas along rivers and streams with clay, sandstone or rock cliffs (Paton 2002). Their core breeding range is found at lower elevations in the Grassland Natural Region (Paton 2002, NRC 2006). Falcons will use a variety of habitats, including grasslands, canyons, cultivated fields, alpine tundra, foothills and dry mountain valleys for foraging and/or breeding (Paton 2002). Areas with extensive forests are avoided. Specialized nesting site requirements make prairie falcons one of the least versatile nesters of Alberta's raptors (Paton 2002). The availability of suitable cliffs, banks or escarpments is a critical breeding habitat requirement for this species.

Similar to the ferruginous hawk, HSI models were developed by MULTISAR to predict prairie falcon habitat use in southern Alberta. The first model was developed for the Milk River Watershed and included three variables: slopes greater than 30°; Richardson's ground squirrel habitat suitability per quarter-section; and distance to ground squirrel habitat (Downey 2004b).

Areas with slopes greater than 30° and with suitable ground squirrel habitat within 15 km of nest sites were given a high suitability rating. Due to the coarse nature of the data, slopes greater than 30° were most representative of steep slopes (*i.e.*, on-the-ground slopes of 75° or greater) within the Milk River Watershed (Downey 2004b). A second HSI model was developed for prairie falcon habitat suitability in the headwaters of the Oldman River in southwestern Alberta (Downey 2004c). It included the same three variables as the first HSI model except that it included slopes greater than 25° due to the fact that a different dataset was used.

Also as part of the MULTISAR program, a Resource Selection Function (RSF) model was developed for prairie falcon habitat use in southern Alberta (Skilnick *et al.* 2010). RSFs are thought to be more robust than HSI models due to their rigorous and empirical evaluation of model performance using statistics (Boyce *et al.* 2002). RSFs estimate relative probability of occurrence by species of concern on the landscape based on various physical and anthropogenic variables. The best-fit RSF model for prairie falcons in southern Alberta, as determined by Skilnick and Dodd (2011), contained 13 variables, including; northness; slope; presence of valley or coulee; distance to badlands; distance to uplands; distance to cropland; percent cover of water; percent shrub cover; amount of edge habitat; distance to fluvial deposits; distance to previous nest locations; and distance to rural and urban areas. Distance to historic nesting sites was considered the most significant factor influencing prairie falcon occurrence (Skilnick and Dodd 2011). Slope did not fully describe falcon habitat use but was considered the second most significant variable. According to the best-fit model, the relative probability of prairie falcon occurrence increases with proximity to fluvial deposits, past nesting sites, rural areas, cropland, native upland, badlands and valleys/coulees. Higher shrub and water cover as well as high amounts of edge habitat also appear to be preferred. The top model also suggests that prairie falcons prefer habitat away from urban areas. The RSF model was significantly better at predicting prairie falcon habitat use than the original HSI model (Downey 2004b, Skilnick and Dodd 2011).

6.3.2 Nesting Habitat

Falcons prefer to nest on secure cliffs with an overhang that provides shelter (Paton 2002). Such sites are typically found along rivers. Nests have been found in natural cavities as well as man-made holes or ledges dug into cliffs of varying substrates, and at a range of heights (Runde and Anderson 1986, Paton 2002). Runde and Anderson (1986) pooled nest site data for 418 sites from numerous studies across the prairie falcon breeding range. They found that the average nest height was 29.3 m from the bottom of the cliff and situated in the upper two-thirds of the cliff. Nests ranged in height from 2.1 m to 154.4 m from the cliff bottom (Runde and Anderson 1986). Prairie falcons have also been known to use abandoned common raven, golden eagle and ferruginous hawk nests (Paton 2002). Occasionally, nests located in trees and transmission towers have been reported (MacLaren *et al.* 1984, Roppe *et al.* 1989). Studies have shown that falcons show fidelity to their breeding territories and return to the same territory used the previous year (Paton 2002). Cliff territories can have more than one nest site, with sites used alternately. Nest sites or nesting territories used repeatedly are called “traditional sites” (Paton 2002). In southern Alberta there are six known traditional sites that have been occupied for a minimum of 10 years, with one nest site used for 23 years (Paton 2002).

6.3.3 Foraging Habitat

Prairie falcon foraging habitat corresponds with habitats that can sustain viable populations of their major prey, Richardson's ground squirrels. Availability of a ground squirrel prey base is therefore thought to be a key factor in the distribution of prairie falcon home ranges in southern Alberta (Hunt 1993, Paton 2002). Hunt (1993) found that radio-tagged prairie falcons along the Bow River hunted as far as 20 km from their nests, but that the birds mostly used native prairie with ground squirrel habitat that was within 15 km of nest sites. Large areas of cropland are thought to have eliminated areas of ground squirrel habitat along the Bow River (Usher 1993). Usher (1993) reported that nesting populations of prairie falcons along the lower Bow River declined by greater than 10% while agricultural land-use within 6 km of the Bow River increased by 25% from 1972 to 1988. Usher (1993) reported that areas of native prairie contained most of the ground squirrel habitat in their study area.

In general, prairie falcon hunting ranges decrease during years of abundant prey, and unlike nest sites, are not vigorously defended (Paton 2002). In southwestern Idaho, Marzluff *et al.* (1997b) reported that prairie falcons that nested near habitat most suitable for Townsend's ground squirrels, the primary prey in the area, ranged over smaller areas and were more successful breeders than falcons that had to range further to locate ground squirrel prey. Marzluff *et al.* (1997b) confirmed the importance of native grassland habitats as key foraging areas, particularly areas with a mosaic of shrubs and grasses. Overall prey abundance was found to be much lower in agricultural lands than in native shrubland in this study. In addition, grassland communities dominated by exotic annuals such as downy brome (cheat grass) (*Bromus tectorum*) appeared to provide less suitable habitat for ground squirrels during drought conditions compared to native perennial grass or shrub mosaics (Marzluff *et al.* 1997b, Steenhof *et al.* 1999). Marzluff *et al.* (1997b) reported that Sandberg's bluegrass (*Poa secunda*), winter-fat (*Krascheninnikovia lanata*) and big sagebrush habitats supported the highest abundances of ground squirrels when their populations were low (Marzluff *et al.* 1997b). Shrub cover is thought to be important to providing forage and cover for ground squirrels, particularly during periods of drought (Marzluff *et al.* 1997b). Interspersed open grassland areas are important as they provide a more effective hunting ground for prairie falcons attempting to catch prey (Marzluff *et al.* 1997b).

Another important component of prairie falcon foraging habitat is the availability of perch sites, such as snags, fence posts, rock faces, utility poles and hay bales (Paton 2002). Perch sites are often used as vantage points from which to seek-out or consume prey (Snow 1974).

6.3.4 Area Requirements

Hunt (1993) estimated the average minimum home range size for prairie falcons nesting along the Bow River to be 72 km^2. Home ranges varied from 31 km^2 to 192 km^2 for the 11 radio-tagged birds in this study (Hunt 1993). Hunt (1993) estimated the core foraging areas for five of the pairs to be 26 km^2 to 40 km^2, with core areas overlapping between adjacent territorial pairs. Marzluff *et al.* (1997b) confirmed that prairie falcons use distinct areas of native grassland within their foraging territories. Native grasslands most likely to contain ground squirrels in this study were located 5 km to 20 km from the nest sites (Marzluff *et al.* 1997b). Marzluff *et al.* (1997b) found that home range sizes increased with declining prey abundance and were larger in areas with reduced cover of perennial grasses. Falcons that ranged greater than 300 km^2 to forage

had less reproductive success than falcons that hunted over areas of between 200 km^2 to 280 km^2 (Marzluff *et al.* 1997b).

7 RAPTOR GROUP COMPARATIVE SUMMARY

Table III-1 provides a comparative summary of the five raptor species considered in this section, including a list of key management concerns for each species. Management considerations are discussed further in Sections 8, 9 and 10.

Overall, habitat loss or alteration is the key limiting factor for all species due to the importance of native prairie for providing either critical foraging or nesting habitat for these raptors. Of note, grazing is the dominant land-use in the majority of the breeding areas used by these raptors in Alberta. As discussed later, grazing may be particularly important for maintaining open habitat for Richardson's ground squirrels, the dominant prey consumed by ferruginous and Swainson's hawks and prairie falcons. Meadow voles, the primary prey consumed by short-eared owls, have fluctuating populations and rely on greater vegetation cover than ground squirrels. White-tailed jackrabbits and mountain cottontails, the primary prey consumed by golden eagles, rely to a greater extent on shrubby cover and edge habitat. All of these prey species are most commonly associated with native prairie. Isolated or small clumps of trees provide important nesting habitat for ferruginous and Swainson's hawks. Cliffs and hoodoos within the Milk River and South Saskatchewan River watersheds also provide nesting sites for ferruginous hawks as well as prairie falcons and golden eagles (Quinlan *et al.* 2003). Short-eared owls are the only strictly ground-nesting raptor. While these raptor species differ in their relative tolerances to human activity, in general, all raptors are most susceptible to human disturbance near nest sites early in the nesting season. The key management concerns listed in Table III-1 are based on a consideration of important limiting factors, dominant prey, primary foraging and nesting habitat and area requirements.

Table III-1 Raptor Group Comparison Summary

Species	Key Limiting Factor	Dominant Prey (in Southern Alberta)	Foraging Habitat (General)	Nest Site Characteristics	Land-use Preference / Distance from Disturbance	Area Requirements	Key Management Concerns
Ferruginous hawk	▪ Habitat loss / alteration ▪ Richardson's ground squirrel abundance ▪ Disturbance during early nesting season ▪ Availability of nesting sites	▪ Primary: Richardson's ground squirrel ▪ Secondary: ▪ white-tailed jackrabbit; ▪ meadow vole; northern pocket gopher	▪ Native prairie with Richardson's ground squirrels	▪ Cliffs along rivers near to suitable foraging habitat ▪ Lone or small clumps of trees or large shrubs; utility structures; dirt outcrops ▪ Ground nests more common where alternative tall structures are absent (often on raised areas in lightly grazed pastures)	▪ Requires at least 50% native prairie ▪ Nests typically located a minimum of 500 m from human disturbance	▪ Average territory size: 2.6 km^2 to 7.7 km^2 ▪ Average hunting distance from nests in Alberta: 800 m	▪ Protect native prairie and Richardson's ground squirrel habitat ▪ Protect nest sites from human disturbance and heavy grazing
Swainson's hawk	▪ Habitat loss / alteration ▪ Richardson's ground squirrel abundance ▪ Nestling survival	▪ Primary: ▪ Richardson's ground squirrel (juveniles) ▪ Secondary: ▪ meadow vole; juvenile white-tailed jackrabbit; mountain cottontail; insects	▪ Native prairie with Richardson's ground squirrels and interspersed smaller blocks of hayland or cropland with vole prey ▪ Cropland and hayland used after mowing or harvest early or late in the season	▪ Trees and shelterbelts often used for nesting; man-made structures used less often ▪ Ground-nesting occasionally reported	▪ Higher nesting densities occur in areas with 11%-30% cultivation ▪ Tolerant of greater amounts of human disturbance than ferruginous hawks and will nest closer to human activity	▪ Home range size: 6.2 km^2 to 27.3 km^2	▪ Protect native prairie and Richardson's ground squirrel habitat ▪ Protect and manage cottonwood forests and shelterbelts
Golden eagle	▪ Habitat loss / alteration ▪ Collisions with vehicles or human structures ▪ Electrocution ▪ Disturbance during early nesting season ▪ Low density, large range	▪ Primary: ▪ white-tailed jackrabbit and mountain cottontail ▪ Secondary: ▪ Richardson's ground squirrel	▪ Shrubby, sagebrush or edge habitats with lagomorph prey	▪ Cliffs along rivers, waterbodies or coulees ▪ Mature, sturdy trees also used for nesting less commonly	▪ Avoids cultivated fields for foraging	▪ Uses core areas ranging from 0.3 km^2 to 15.35 km^2 during the breeding season for foraging	▪ Protect lagomorph prey habitat ▪ Protect tree and cliff nest sites ▪ Minimize disturbance at nest sites

Short-eared owl	▪ Habitat loss / alteration ▪ Meadow vole abundance ▪ Heavy grazing over large areas ▪ Egg predation	▪ Primary: ▪ Meadow vole ▪ Secondary: ▪ other *Microtus* (vole) species; shrews; pocket gophers; mice; kangaroo rats	▪ Various grassland habitats with meadow vole prey	▪ Ground nests usually in native prairie (ungrazed or lightly grazed)	▪ Uses a variety of native and non-native grassland habitats where vole populations are abundant ▪ Dense grassland	▪ Average breeding territory in Manitoba: 0.82 km^2 ▪ Breeding territory size varies with fluctuating vole densities	▪ Protect meadow vole habitats ▪ Minimize disturbance to ground nests and provide adequate nesting cover ▪ Lightly graze or rest pastures with active nests
Prairie falcon	▪ Habitat loss / alteration ▪ Richardson's ground squirrel abundance ▪ Availability of nesting sites	▪ Primary: ▪ Richardson's ground squirrel ▪ Secondary: ▪ grassland passerines	▪ Native prairie with Richardson's ground squirrels adjacent to nesting areas	▪ Cliffs along rivers and waterbodies near to suitable foraging habitat	▪ Avoids cultivated fields for foraging	▪ Average home range along Bow River site: 72 km^2; most hunted within 15 km of nests	▪ Protect cliff nest sites from disturbance ▪ Protect native prairie and ground squirrel habitat

8 GRAZING AND RAPTORS

As described by Kochert *et al.* (1988) and Kochert (1989), grazing has the potential to influence raptors by affecting three factors in particular, including the:

- Quality and availability of nesting substrate.
- Diversity and abundance of prey.
- Vulnerability of prey to raptor predation by removal of cover.

These three factors ultimately influence the reproductive success and density of raptors (Kochert *et al.* 1988). The ability to quantify these grazing effects is difficult and to date no definitive quantitative studies have been completed (Olendorff 1993). A discussion of the effects of grazing on the nesting substrate and prey base of the five raptors considered in this section is given below. The influence of various grazing systems on each of these factors is discussed further in Table III-2.

1) Modification of Tree, Cliff and Ground-nesting Substrates

The availability of suitable, safe nesting sites has obvious implications for the reproductive success of raptors. The potential effects of grazing on ground-, tree- and cliff-nesters are discussed below.

A) Tree Nests

Tree-nesting species include ferruginous and Swainson's hawks and, to a lesser extent, golden eagles and prairie falcons. These species all make use of lone or small clumps of trees in upland areas and riparian habitats. Mature cottonwood forests commonly found along streams and rivers in southern Alberta offer particularly important nesting habitat for raptors and numerous other birds. In southwestern Alberta, plains cottonwood, balsam poplar (*Populus balsamifera*), narrowleaf cottonwood (*P. angustifolia*) and interspecific hybrids provide the foundation of biologically diverse riparian forests (Gom and Rood 1999). In general, excessive trampling and repeated browsing of mature trees or seedlings stunts growth and limits the survival and regeneration potential of trees. This type of impact reduces the capacity of trees to provide a suitable and stable nesting substrate. Bradley and Smith (1986) contend that heavy browsing and trampling by cattle negatively affects cottonwood regeneration and density along the Milk River. Cottonwood trees are, however, ultimately dependent on sufficient instream flows and flooding events for growth and seed dispersal and germination (Bradley and Smith 1986). Sparse trees in arid landscapes are especially susceptible to heavy use from cattle if alternate shade or rubbing structures are not available. A lone tree or small clump of trees may be killed through repeated rubbing by cattle causing girdling of the trunk.

The timing and intensity of grazing, placement of upland watering and salt sources and frequency and duration of rest periods will have an effect on how severely cattle impact trees. Appropriate riparian area management is particularly important for limiting impact to trees. Fencing may be required to protect upland nest trees in arid landscapes that are consistently

heavily impacted by livestock. This is likely to be more of a concern under continuous (season-long) grazing.

B) Cliff Nests

Golden eagles and prairie falcons nest primarily in cliff habitats in southern Alberta. Cliff and hoodoo sites along the upper Milk River (including the North Milk River) are also commonly used for nesting by ferruginous hawks (Erickson 2000, Quinlan *et al.* 2003, Downey and Quinlan 2004). As these nest sites are usually inaccessible to cattle, direct impacts to cliff nests are minimal. However, heavy use of riparian areas by cattle could have long-term indirect effects on cliff nest-site stability. Heavy trampling and browsing along riparian areas can lead to removal of woody vegetation and an altered plant community comprised of shallow-rooted exotic grasses (Fitch and Adams 1998). This type of severe modification can cause erosion and bank slumping, which eventually alters channel morphology and can lead to increased velocity of streams and rivers (Fitch and Adams 1998). This has downstream consequences on bank stability and can result in increased erosion, which ultimately leads to slumping, incisement and/or possible flooding of cliff-nest sites. Nest sites nearer to the base of cliffs are more susceptible to this type of damage. Although these types of effects are typically the result of other human impacts at the watershed level (*i.e.*, diversion of water, improper culvert sizes, damming or channeling of natural watercourses) cumulatively, localized impacts due to grazing at several points along the same riparian channel can have a significant effect.

C) Ground Nests

Short-eared owls are one of the few prairie raptors that nest predominantly on the ground. Ferruginous hawks, Swainson's hawks and golden eagles will also nest on the ground when suitable elevated structures are not available (Snow 1973, Dechant *et al.* 2002a,b). Ground nests are inherently susceptible to the direct and indirect consequences of grazing. Heavy grazing can reduce the availability of suitable grass cover required for nesting, particularly if stocking rates do not allow for sufficient carry-over of litter each year and do not account for periods of drought. Ground nests may also be susceptible to trampling, particularly in intensively stocked pastures. As discussed earlier, ferruginous hawk ground nests in South Dakota were preferentially located in prairie that was unused or lightly grazed (Lokemoen and Duebbert 1976). Short-eared owls have also demonstrated a preference for nesting in lightly grazed or ungrazed fields (Dechant *et al.* 2002c).

2) Alteration of Prey Diversity and Abundance

In general, small mammals show a varying degree of tolerance towards grazing (Kochert *et al.* 1988, Kochert 1989, Olendorff 1993). Grazing typically favours granivorous small mammals such as heteromyid (pocket mouse) and geomyid (pocket gopher) rodents; however, few prey species tolerate intensive long-term grazing (Kochert *et al.* 1988, Kochert 1989). Jones *et al.* (2003) suggest that a landscape with a mosaic of grass and shrublands and varying amounts of ground cover, including relatively dense grasslands, will likely maintain the highest diversity of rodents.

The habitat preferences and potential impacts of grazing on the primary prey species listed in Table III-1 are discussed below.

A) Richardson's Ground Squirrels

Richardson's ground squirrels are common throughout the Grassland Natural Region of Alberta (NRC 2006). Ground squirrels are found in colonies and prefer flatter, upland, open areas of short and mixedgrass prairies with relatively short vegetation that allows them to detect predators (Michener 1996, 2002, Reynolds *et al.* 1999, Michener and Schmutz 2002). Soil type is a limiting factor as to where ground squirrels occur. Ground squirrels do not inhabit areas with loose sand or heavy clay soils (Reynolds *et al.* 1999). Richardson's ground squirrels are apparently able to survive in human-modified habitats, such as in heavily grazed fields, at the edges of cultivated fields, perennial crops and in green spaces in urbanized areas (Michener 2002). However, according to a ground squirrel survey in southern Alberta, native prairie appears to be the preferred habitat type (Downey 2003b). Out of a total of 796 ground squirrels counted during this survey, 571 (72%) were found on native prairie (n = 696); 128 (16%) were found in cultivated fields (n = 808); 58 (7%) were observed in tame pasture (n = 141); 32 (4%) were observed in hay land (n = 15); and 7 (0.9%) were observed in farmyards (n = 24). The majority of ground squirrels found in native prairie were observed in short vegetation (*i.e.*, less than 10 cm). Although grazing use was not quantified during this study, these results indicate that ground squirrels appear to tolerate moderate to heavy grazing that results in short vegetation.

B) White-tailed Jackrabbits and Mountain Cottontails

White-tailed jackrabbits are common throughout southern and central Alberta and typically occur in open areas, avoiding dense forests (Pattie and Fisher 1999). Jackrabbits prefer upland habitats comprised of open grassland with intermittent shrubby cover (Reynolds *et al.* 1999). Mountain cottontails occur throughout the Grassland Natural Region of Alberta (Pattie and Fisher 1999, NRC 2006). Their range does not extend beyond the northern limit of the Red Deer River. Unlike white-tailed jackrabbits, mountain cottontails avoid open, coverless plains and prefer wet, lowland areas, such as cottonwood tree groves and shrubby ravines (Reynolds *et al.* 1999). Therefore, cover and edge habitat are described as important habitat features of mountain cottontails in Alberta (Pattie and Fisher 1999).

Flinders and Hansen (1975) found that white-tailed jackrabbits were not significantly affected by grazing intensity in the shortgrass prairie of northeastern Colorado. However, desert cottontail rabbits were most abundant in moderate-summer and moderate-winter grazing treatments in this study. Desert cottontail rabbits were most common in areas near dense stands of shrubs, along edges between vegetation types or in areas with gullies or rocky outcrops. In Oklahoma, Great Plains jackrabbits (*Lepus californicus melanotus*) and pocket gophers (*Geomys breviceps llanensis*) were found to be more abundant in "moderately overgrazed" pastures, whereas Oklahoma cottontails (*Sylvilagus floridanus alacer*) were most abundant in undisturbed grasslands and least abundant in heavily grazed pastures (Philips 1936). Overall, rodent and lagomorph numbers were reduced in "heavily overgrazed" pastures in this study. In southern British Columbia, sagebrush habitats with at least 30% vegetative cover represented good habitats for mountain cottontails (Sullivan *et al.* 1989).

C) Meadow Voles

Meadow voles occur throughout Alberta in a variety of habitats (Pattie and Fisher 1999). The number and survival rates of meadow voles have been found to increase with increasing grass cover (Adler and Wilson 1989, Peles and Barrett 1996, Reynolds *et al.* 1999, Lin and Batzli 2001). Above-ground herbaceous biomass of 700 g/m^2 is sufficient to support high densities of meadow voles; however, vole densities decline rapidly when cover values drop to less than 280 g/m^2 (Reynolds *et al.* 1999). Due to this preference for high amounts of vegetation cover, heavy grazing is likely detrimental to meadow voles. Unlike cottontails, studies have also shown that meadow voles are less abundant along wooded edges and increase in number in the prairie interior in grass and forb dominated habitats (Manson *et al.* 1999, Nickel *et al.* 2003). Within the Milk River area, meadow voles were found to rely to some extent on the tunnels created by Richardson's ground squirrels, particularly in areas with low cover values (Salt 2000).

3) Changes in Prey Vulnerability

By reducing vegetation cover, grazing may improve the ability of raptors to detect and catch prey. Wakeley (1978) and Bechard (1982) discuss the effect of vegetative cover on the use of foraging sites by Swainson's hawks and ferruginous hawks, respectively. Wakeley (1978) and Bechard (1982) both found that the density of vegetation cover was more important than prey density in the selection of hunting sites by Swainson's and ferruginous hawks. Bechard (1982) found that Swainson's hawks did not hunt in cultivated fields until crop harvest, despite an abundance of prey in cultivated fields prior to harvest. In general, densely covered cropland offers better concealment of prey, which thereby reduces prey availability and limits foraging efficiency. By concentrating hunting effort in areas with reduced vegetation cover and density, hawks are able to gather more food per unit of energy expended (Wakeley 1978). This theory explains why native prairie, in comparison to dense cropland, provides better quality foraging habitat to all prairie raptors. In addition, it can explain how grazing, by reducing vegetation cover and density, may offer improved foraging success to raptors by increasing the vulnerability of prey. However, these positive effects depend on the intensity and duration of grazing as prolonged heavy grazing can eventually lead to decreased species diversity and abundance of certain prey species, such as meadow voles (Kochert *et al.* 1988).

9 GRAZING SYSTEMS AND RAPTOR HABITAT MANAGEMENT

Table III-2 provides an overview of five pertinent grazing systems and their potential positive and negative effects on raptors and their habitat. A grazing system is a tool used to control the spatial distribution, timing, intensity, and frequency of livestock grazing (Holechek *et al.* 2003). Applied research is needed to properly assess the effects of various grazing systems on raptors in the Milk River and South Saskatchewan River watersheds.

Table III-2 Grazing Systems and Raptor Habitat Management

Grazing System	Discussion
Continuous (Season-long) Grazing	
Advantages:	The potential advantages of continuous grazing systems to raptors depend on stocking rate, timing of grazing and the effectiveness of techniques to distribute grazing use over the landscape. With light to moderate stocking rates, areas near water or salt or with more palatable grasses will receive heavier use than other areas. Formerly grazed patches will receive repeated use as these patches have a lesser build-up of litter and higher cover of more palatable re-growth vegetation (Robertson *et al*.1991). Therefore, while cattle are not excluded from certain areas under continuous grazing, with stocking rates that ensure that only a moderate percentage of vegetation is utilized, grazing pressure across the rangeland will be variable. Variably grazed rangeland creates patches with varying litter cover and vegetation heights and also stimulates plant species diversity (Bai *et al.* 2001). Richardson's ground squirrels will benefit from shorter vegetation in moderate to heavily grazed patches (Downey 2003a, Downey 2004a), while lagomorph prey and meadow voles will benefit from the greater vegetation cover of lighter or ungrazed areas (Olendorff 1993). Therefore all primary raptor prey species can benefit from heterogeneous cover. In areas with less cover, prey vulnerability is increased, allowing raptors to forage more efficiently (Wakeley 1978, Bechard 1982). Areas with less use and higher cover also offer protected ground-nesting sites.
Disadvantages:	In the long-term, continuous grazing under heavy stocking rates has the potential to negatively impact foraging and nesting habitat for most raptors. Short-eared owls, as ground- nesters and whose primary prey (meadow voles) prefer greater amounts of cover, may see immediate consequences of heavy grazing. Bock *et al*. (1993) suggested short-eared owls responded negatively to grazing in the Great Plains and western shrubsteppe. Other studies suggest short-eared owls require idle (ungrazed during the breeding season) grasslands or dense, tall vegetation for nesting (Duebbert and Lokemoen 1977, Kantrud and Higgins 1992, Dechant *et al.* 2002c). Raptors such as Swainson's and ferruginous hawks that rely on nest trees in riparian areas may also be negatively affected in the long-term by continuous heavy grazing. Riparian areas tend to suffer the most under continuous heavy use, particularly if alternate water sources or upland salt sites are not available. Heavy use of riparian areas leads to trampling and heavy browsing and ultimate elimination of woody species required for nesting. Heavy use of riparian area vegetation also limits their potential to provide habitat for numerous other birds and small mammals that rely on dense or woody cover and a diverse plant community.

Complementary Grazing	
Advantages:	Complementary grazing is a form of season-of-use grazing, with tame pastures grazed earlier or later in the season than native prairie. The use of tame pasture has the potential to improve the health of native rangeland by allowing periods of rest and recovery. Improved rangeland health allows for retained and increased litter cover and restored plant vigour of upland grassland as well as riparian areas and woody draws. This has potential to improve the availability or sustainability of nesting cover and prey habitat for raptors. The benefits of complementary grazing, however, depend on whether marginal cropland can be converted to tame pasture or if tame pasture is available within the grazing operation. Interspersed tracts of tame pasture is likely most beneficial to Swainson's hawks, as they use a variety of cover types for foraging and have a broader diet than ferruginous hawks and prairie falcons (Dechant *et al.* 2002b).
Disadvantages:	Conversion of large tracts of native prairie to monotypic tame pasture has the potential to reduce the density and reproductive success of raptors through the loss of nesting sites and possible reduction in the diversity or availability of prey populations (Howard and Wolfe 1976). Although further study is required, a recent survey suggests that Richardson's ground squirrels are significantly more common in native prairie than cultivated land or tame pasture in southern Alberta (Downey 2003a). Therefore loss of native prairie has direct implications for the availability of primary prey for ferruginous and Swainson's hawks and prairie falcons. Ferruginous hawks may be particularly affected by fragmentation of native prairie as they decline in abundance in areas with less than 50% native prairie cover (Schmutz 1989). Similarly, prairie falcons have been found to hunt primarily in native prairie (Hunt 1993). Monocultures may also have a negative effect on golden eagles if no patches of shrubby or woody cover vegetation are retained to support lagomorph prey species.

Rotational Grazing – Rest Rotation and Deferred Rotation	
Advantages:	Deferred or rest-rotational grazing systems have been recommended as good tools for managing raptor habitat in general (Kochert *et al*. 1988, Kochert 1989). These systems have been specifically recommended for ferruginous hawks (Ensign 1983, Olendorff 1993) and short-eared owls (Kantrud and Higgins 1992, Herkert *et al.* 1999). Rotational grazing systems are preferred over other types of grazing strategies as they offer a means to create habitat diversity by enforcing periods of rest and recovery. Ensign (1983) recommended lower levels of grazing under rest-rotation systems for managing ferruginous hawk habitats. Ensign (1983) suggested that rotational grazing systems create diversity by creating a distribution of heavy, moderate, light and ungrazed pastures. Moderate and heavily grazed pastures increase the vulnerability of prey, while ungrazed or lightly grazed pastures provide refuge habitat and a stable prey population source. Ensign (1983) recommended low to moderate grazing levels to ensure sufficient food and cover for prey species, particularly during drought years. Kantrud and Higgins (1992) found that fields that were idled during the growing season had more short-eared owl nests than fields under long-term grazing. Skinner *et al.* (1996) found that deferred grazed native prairie in southeastern Alberta had the highest values for vegetative cover and were the most productive habitat type for small mammals. Another benefit of rotational grazing is that it offers a tool for riparian habitat management by being able to control the timing and intensity of use in riparian areas. Riparian habitat management is important for protection of nest trees and to prevent severe channel modification and potential impact to cliff nest sites. Periodic rest also allows for improved health of upland shrub or tree vegetation. Structurally, as rotational grazing requires the use of cross fencing, it offers numerous perch sites for raptors to use for territory defense or hunting.
Disadvantages:	If cattle are grazed beyond the 50% utilization point, rotational grazing can contribute to uniform grazing effects (Kobriger 1980). This is particularly common when rotation of cattle between pastures is based on set calendar dates rather than according to vegetation utilization. Uniform utilization of the range can have short-term benefits to raptors by increasing the vulnerability of prey to predation and possibly by increasing the density of Richardson's ground squirrels. However, it does not encourage the heterogeneity that is required to maintain stable and diverse small mammal prey populations or offer protected ground or tree nesting sites.

Rotational Grazing – High Intensity Grazing	
Advantages:	High stocking densities for short periods encourages intense utilization of the range, forcing cattle to be less selective when grazing. By distributing animals into numerous small grazing units, there is rigid control over where and when cattle graze portions of the landscape. Following periods of intense grazing, the resulting shorter vegetation offers raptors the benefit of increased prey vulnerability. Short vegetation also creates conditions that may be favourable for Richardson's ground squirrels, which benefit from improved visibility and ability to detect predators (Michener 2002, Downey 2003a, Downey 2004a). Certain grasshopper species also flourish in more heavily grazed rangeland, offering an alternate prey source for Swainson's hawks in particular (Johnson *et al.* 1987). Holmes *et al.* (1979) collected more grasshoppers from heavily and very heavily grazed fescue grasslands than from lightly and moderately grazed lands. Similarly, Quinn and Walgenbach (1990) found that grazed mixedgrass prairie sites supported higher populations of obligate grass-feeding grasshoppers. As intensive grazing systems require the creation of numerous fenced units, this creates numerous perches for raptors.
Disadvantages:	Intensive grazing, even for short periods of time, can result in declines in range condition (health), lower root mass, lower vegetation densities, soil compaction and reduced infiltration rates in mixedgrass and fescue prairie (Dormaar *et al.* 1989, Willms *et al.* 1993). Reduced vegetation cover results in a decrease in nesting cover for short-eared owls as well as reduced cover for their primary prey, meadow voles. Ground-nesting cover in areas with sparse tall nesting structures is also reduced for ferruginous and Swainson's hawks. In South Dakota, ferruginous hawks preferred to nest in lightly grazed or idled areas (Dechant *et al.* 2002a). Reduced cover also diminishes the quality of habitat for jackrabbits and cottontails. As grazing is non-selective under intensive grazing systems, upland and riparian woody vegetation is typically browsed more heavily than it would be with reduced stocking rates. Heavy browsing, rubbing and trampling weaken nest trees. Therefore, intensive grazing has evident short- and long-term detriments to raptors particularly if periods of rest are insufficient.

Riparian Area Grazing	
Advantages:	In the dry prairies of southern Alberta, suitable nesting trees for golden eagles, Swainson's and ferruginous hawks are most likely to be found in riparian areas along streams, rivers or lakes. Riparian areas that are well-managed through either rest-rotation, corridor fencing or riparian pasture systems are more likely to support healthy, diverse vegetation communities with deeply-rooted woody species and large shrubs for nesting. Healthy and diverse riparian vegetation also provides habitat for a diversity of small mammal and insect prey.
Disadvantages:	Other than mountain cottontails, the prey species listed in Table III-1 are predominantly found in open, grassland habitats. Thus, protection of riparian habitats should not compromise upland forage or cover for these species or diminish nesting cover for ground-nesting raptors.

10 BENEFICIAL MANAGEMENT PRACTICE RECOMMENDATIONS

Protection of prairie raptors, including ferruginous hawks, Swainson's hawks, golden eagles, short-eared owls and prairie falcons, is contingent on maintaining suitable and secure (*i.e.*, undisturbed) nesting habitat surrounded by good quality foraging habitat with an adequate and stable prey base. Although there is some variation in the nesting and dietary requirements of these raptors, stewardship of native prairie intermixed with shrubby draws and tree clumps as well as healthy riparian corridors will benefit all species.

The following general land-use and grazing recommendations offer a variety of means to protect or enhance raptor habitat within the Milk River and South Saskatchewan River watersheds and throughout the Grassland Natural Region of Alberta (NRC 2006). Further research is required to improve our understanding of prairie raptors and their prey as well as the role of grazing as a tool for enhancing raptor habitat (see Section 11).

10.1 General Recommendations

Foraging Habitat

- Protect remaining native prairie from cultivation (Olendorff 1993, Schmutz 1999, Dechant *et al.* 2002a,b,c). In particular, maintain native prairie complexes within 20 km of major river valleys with suitable cliff nest sites for ferruginous hawks, golden eagles and prairie falcons (Hunt 1993, Quinlan *et al.* 2003). Ferruginous hawks require large areas of contiguous native prairie, while other raptors, such as Swainson's hawks, will use fragmented native prairie with between 11% and 30% cultivation (Schmutz 1984). Land managed for cattle grazing is compatible with the protection of large tracts of native prairie.
- Remove marginal farmland from production, where possible, and seed with native species to enlarge habitat patches and reduce fragmentation for ferruginous hawks in particular. Work with a qualified Agrologist, Biologist or reclamation/restoration specialist to ensure appropriate species selection and seeding techniques are used.
- Reduce or avoid the use of insecticides and organochlorine pesticides, including carbamates and organophosphates (Forsyth 1993, Baril 1993, Kirk and Hyslop 1998). Organochlorine pesticides such as carbofuran are considered extremely toxic to birds (Baril 1993). Raptors are especially vulnerable to poisoning through ingestion of contaminated prey carcasses (Baril 1993). The widespread use of organochlorine pesticides may also negatively affect raptor breeding success (Baril 1993).
- In areas with nesting populations of golden eagles or ferruginous hawks, maintain existing native shrub and sagebrush (*Artemisia* spp.) communities (Howard and Wolfe 1976, Olendorff 1993, Marzluff *et al.* 1997a). Large patches of shrub or sagebrush vegetation increases suitable lagomorph (rabbits and hares) prey habitat for these raptor species (Marzluff *et al.* 1997a). Howard and Wolfe (1976) recommend leaving at least 20% of the total area in existing shrub, sagebrush or tree habitat to maintain foraging and breeding habitat for ferruginous hawks.

- Conserve large, open grassland areas near to traditional ferruginous hawk, prairie falcon and short-eared owl nest sites (Holt and Leasure 1993, Olendorff 1993, Herkert *et al.* 1999, Dechant *et al.* 2002a,c).
- Maintain or restore the health of lotic (flowing water) and lentic (standing water) riparian habitats that support tree or shrub vegetation.
- Avoid ground squirrel control within a minimum of 15 km of raptor nests. Inform landowners about the beneficial ecological role of Richardson's ground squirrels and about preferred control techniques if control of this species is considered essential. If control is essential to protect agricultural values, lower the peaks of cyclic highs rather than eliminating entire local populations (Olendorff 1993). Strongly promote raptors as natural biological control agents.
- Retain fences and other tall, man-made structures on the landscape where possible. Such structures provide perching sites for raptors to hunt from.

Nesting Habitat

- Protect traditional raptor tree and cliff nest sites from human disturbance and heavy grazing (Olendorff 1993, Dechant et al. 2002a,b, Paton 2002). Security of nest sites is particularly important for prairie falcons due to their strong nest site fidelity and specialized nesting requirements (Paton 2002). Particular attention should be paid to protecting the North Milk and Milk Rivers from the U.S. – Canada border downstream to the Town of Milk River from development or increased human disturbance (Quinlan et al. 2003, Downey and Quinlan 2004). These rivers have significant nesting populations of ferruginous hawks and prairie falcons (Quinlan et al. 2003, Downey and Quinlan 2004).
- Use Protective Notations under the *Public Lands Act* to conserve key raptor nesting habitat and surrounding foraging habitat.
- Minimize human disturbance near known raptor nest sites during the early nesting period (*i.e.*, March 15 to July 15). This is especially important for ferruginous hawks due to their documented sensitivity to disturbance (Ensign 1983, Olendorff 1993, Dechant et al. 2002a). Increased human activities around river breaks and on the river which may disturb or displace nesting raptors is a particular concern for cliff nesters like prairie falcons, golden eagles and ferruginous hawks. "Wildlife Control Areas" (i.e., public exclusion areas) have proven to be an effective tool for protecting prairie falcon and golden eagle nest sites along the Oldman River Reservoir (Paton 2002).
- Abide by set-back distances and timing restrictions recommended by AEP for human activities, including industrial development, near to raptor nests (GoA 2011). For ferruginous hawk, golden eagle and prairie falcon nests, AEP recommends a 1,000 m set-back for low, medium and high disturbance activities between March 15th and July 15th. For the rest of the year (*i.e.*, between July 16th and March 14th), AEP recommends 50 m, 100 m and 1000 m set-backs for low, medium and high disturbance activities. Short-eared owl nests and surrounding habitat have a recommended set-back of 100 m for all levels of disturbance between April 1st and July 15th.
- Conduct pre-development wildlife surveys to locate ferruginous hawk, Swainson's hawk, prairie falcon, golden eagle and short-eared owl nests to plan developments in accordance

with AEP set-back guidelines. Surveys need only be conducted where suitable habitat for these species exists.

- Conduct land improvements such as brush removal, plowing or mowing during the non-nesting season (*i.e.*, avoid these types of activities between March 1 and August 1) (Olendorff 1993).
- Maintain individual trees or interspersed tree stands in the landscape (Olendorff 1993, Dechant et al. 2002a,b,). The availability of suitably spaced trees affects the number of breeding buteos, such as ferruginous and Swainson's hawks. In Alberta, at least four small clumps of trees per 2.6 km^2 of habitat are necessary to maximize nesting by the majority of buteos (Schmutz et al. 1980, Olendorff 1993).
- Leave dead or decadent trees standing (Olendorff 1993).
- Monitor the condition of nest trees and, where necessary, plant trees or reinforce dead or decadent nest trees (Olendorff 1993). Olendorff (1993) recommends reinforcing the bases of weak or unstable tree nests with wire mesh, netting or lumber as well as stabilizing or reinforcing snags that hold nests.
- Maintain shelterbelts around abandoned farmsteads and field edges for use by Swainson's hawks (Dechant *et al.* 2002b). Replace dead or dying trees along shelterbelts with native species such as cottonwoods.
- Educate landowners on the importance of naturally-occurring trees and shrubs in the prairie ecosystem and their importance to ferruginous haws (AFHRT 2009). Through available stewardship programs, encourage the protection and management of these trees and shrubs.
- Maintain or restore the health of riparian forests. Cottonwood forests along the Milk River provide nesting habitat for golden eagles and Swainson's hawks in particular as well as numerous other bird species. Although impacted by grazing, cottonwood forests require sufficient instream flow particularly during late spring and summer as well as regular flooding events for seed dispersal and germination (Bradley and Smith 1986, Rood *et al.* 1995).
- Where natural cliff nest sites are jeopardized either by natural factors or human activities, investigate the potential to create artificial nest sites for prairie falcons (Paton 2002). Small holes can be excavated in clay cliffs within 1 km of known ground squirrel populations to provide suitable nesting sites for prairie falcons (Paton 2012).
- Artificial cliff nest sites constructed for prairie falcons as part of the mitigation program for the Oldman River Reservoir were successfully occupied (Fyfe 1990). Artificial nest sites have also been successful at the Bassano Reservoir (Paton 2002).
- Where feasible, construct artificial nesting sites for raptors (AFHRT 2009). Human structures such as power lines or nesting platforms are frequently used for nesting by ferruginous hawks in particular (Olendorff 1993). Ferruginous hawk nest pole construction techniques are discussed by Migaj *et al.* (2011). Artificial nesting structures should not be considered substitutes for natural nesting sites.
- Use the GVI database to identify high potential raptor nesting sites (*e.g.*, Badlands/ Bedrock ecological sites adjacent to upland native prairie).

Other

- Various techniques have been suggested to reduce the risk of electrocution deaths among golden eagles in particular. These include promoting the use of low-profile and enclosed transformers; providing a minimum of 2.1 m (7 ft.) of space between phase conductors and ground wires on all lines; and installation of perches at poles where multiple bird deaths have occurred (Snow 1973). Electrical structures in high priority areas can be retrofitted to minimize raptor electrocution mortalities using methods developed by the Avian Power Line Interaction Committee (APLIC 2006, 2012, AFHRT 2009). Raptor collisions with power lines can be reduced through the development of site-specific Avian Protection Plans by utility companies, which provide step-by-step procedures on how the companies will minimize avian deaths as a result of power line collisions/electrocutions (APLIC 2012).
- Develop a provincial raptor monitoring program. Such a program has been suggested for prairie falcons in order to monitor population trends, including occupancy and productivity, over time (Paton 2012). Monitoring strategically-selected sites every two years or conducting a full inventory every five years may be economically feasible. Ensure all observations from the monitoring program (including 'zero data') are submitted for inclusion in the Fish and Wildlife Management Information System (FWMIS) maintained by AEP.

10.2 Grazing Recommendations

Grazing systems that will benefit raptors should perform the following functions:

- Maintain or enhance the quality and availability of nesting substrate.
- Increase the vulnerability of prey to raptor predation while maintaining a diversity of cover types to promote the diversity, abundance and long-term stability of prey populations.

Selection of a suitable grazing system is dependent on local environmental conditions and the types of raptors that are being managed for. Appropriate grazing systems should be developed site-specifically for participating ranches in the MULTISAR conservation program. Grazing strategies that promote heterogeneous vegetation cover and retain native prairie landscapes with mosaics of shrub or tree patches will benefit a wide range of prairie raptors. Creating heterogeneity across the landscape requires control of timing of grazing, stocking rates and cattle distribution.

The following grazing recommendations discuss key management principles that will benefit all prairie raptors. Specific recommendations for individual species are also given.

- Promote heterogeneous vegetation heights by encouraging light to moderate grazing at suitable safe-use factors for native prairie (*i.e.*, 25% to 50% utilization) (Adams *et al.* 2005, 2013a,b).
- Use appropriate stocking rates and safe-use factors to ensure sufficient carry-over and maintenance of plant vigour and range and riparian health.

- Track annual grazing capacities and adjust stocking rates and distribution in accordance with drought events and range and riparian health.
- Patch grazing, which leads to the creation of a mix of heavily grazed and more lightly or moderately grazed patches, will benefit the greatest number of raptor prey species. Ground squirrel density and vulnerability is increased in shorter vegetation (Downey pers. comm., Michener and Schmutz 2002). Taller cover provides preferred habitat for lagomorph prey and for meadow voles, the primary prey of golden eagles and short-eared owls, respectively.
- Avoid intensive grazing systems with high density stocking rates over large areas and retain areas with taller vegetation cover (20 cm to 40 cm in height) to provide ground-nesting habitat for short-eared owls, ferruginous hawks and Swainson's hawks (Holt and Leasure 1993, Herkert *et al.* 1999).
- Use deferred or rest rotational grazing to improve ground-nesting habitat for short-eared owls and ferruginous hawks, in particular (Ensign 1983, Kochert *et al.* 1988, Kochert 1989, Kantrud and Higgins 1992, Olendorff 1993, Herkert *et al.* 1999).
- Manage and monitor cattle use of upland woody vegetation to ensure trees and shrubs are healthy and capable of regenerating. Upland shrub patches provide lagomorph prey habitat, while trees provide nesting habitat for many raptor species. Temporary fencing may be needed to mitigate heavy use of nest trees in arid landscapes with scarce woody cover (Olendorff 1993).
- Implement riparian grazing systems to maintain diverse vegetation cover for small mammal prey species and to maintain varied age classes of trees to provide long-term nesting opportunities for raptors. The optimal time of use of riparian areas is during the summer after spring runoff when the streambanks are no longer soft, and before the dormant season (Fitch and Adams 1998). This time period also reduces disturbance near raptor nests early in the nesting season.
- Distribute salt away from riparian areas and shrubby draws to reduce impact to these areas and encourage better utilization.
- Develop upland stock water, where necessary, to control heavy use of riparian areas.
- Ensure cattle access trails along riverbanks do not impact ferruginous hawk, prairie falcon or golden eagle nest sites along slopes.
- Deferred-rotation, rest-rotation, riparian pasture and corridor fencing have been suggested as techniques for improving the health of riparian areas (Fitch and Adams 1998). Several years of rest may be required where the goal is to regenerate new trees such as cottonwoods (Fitch and Adams 1998).

11 RESEARCH RECOMMENDATIONS

Few quantitative studies have assessed the effects of grazing and different grazing strategies on raptors and their habitats (Olendorff 1993). Continued monitoring of raptor populations, their primary prey and changes in land-use and grazing intensity over time is necessary. This type of monitoring is currently ongoing with respect to ferruginous hawks and Richardson's ground squirrels in the Grassland Natural Region of Alberta (Downey 2003a, Taylor 2003, NRC 2006). However, similar research is lacking for prairie falcons, short-eared owls and golden eagles

(Clayton 2000). There is a need for improved survey protocols and more consistent monitoring of core populations of these species to provide a reliable index of population and productivity trends (Paton 2002). Foraging habitat use by short-eared owls and golden eagles in Alberta also requires further study. Lastly, the impact of climate change on raptor species and their prey has yet to be studied.

With respect to grazing and raptors, specific questions to be addressed include:

- Is there a significant difference between raptor densities or reproductive success in habitats managed under continuous versus rotational grazing systems?
- How does grazing intensity affect raptor prey species composition, density and long-term stability?
- What is the frequency of foraging attempts by raptors in grazed versus ungrazed habitats?
- How does range or riparian health correspond with raptor productivity or density?

Due to their importance as a major prey species for the majority of raptors, additional scientific research to evaluate the habitat preferences of Richardson's ground squirrels would be valuable (Michener pers. comm.). Little is known about the response of ground squirrels to habitat fragmentation or the relative stability and abundance of populations in shorter versus taller vegetation. Additional research is also required to determine how many ground squirrels are needed to support a productive population of prairie falcons (Paton 2002). The relative importance of ground squirrels as secondary prey for golden eagles also requires further study.

Paton (2012) listed the following research objectives for prairie falcons in Alberta, which can be extrapolated to include all raptor species in the province:

- Investigate habitat use by prairie falcons during the summer, fall and post-breeding periods.
- Determine habitat characteristics for both declining and stable prairie falcon subpopulations at multiple scales.
- Evaluate the genetic exchange between prairie falcon subpopulations.
- Research post-fledgling survival and dispersal success of prairie falcons.

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B. SHARP-TAILED GROUSE

1 INTRODUCTION

The purpose of this report is to summarize the ecology and critical habitat needs of the plains sharp-tailed grouse (*Tympanuchus phasianellus jamesi*) in southern Alberta. Based on this information and supporting scientific studies, the potential effects of livestock grazing on this species and its habitat is discussed. This discussion is followed by a summary of recommended beneficial management practices to enhance sharp-tailed grouse habitat in the Milk River and South Saskatchewan River watersheds in Alberta, with broader application to the range of this species within the Grassland Natural Region of Alberta (Natural Regions Committee 2006). Lastly, a brief summary of research recommendations is provided.

1.1 Background

Sharp-tailed grouse are one of five grouse species in Alberta. Sharp-tailed grouse are medium-sized grouse with yellowish-orange eye combs and sharp-pointed tails (Semenchuk 1992). The species occurs throughout much of Alberta where suitable habitat exists, including five of six natural regions (Semenchuk 1992; Federation of Alberta Naturalists 2007). Sharp-tailed grouse are most common in the Grassland and Parkland natural regions (Federation of Alberta Naturalists 2007).

Sharp-tailed grouse are currently listed as ‘Sensitive’ in Alberta under the General Status listing (Government of Alberta [GoA] 2016). The sharp-tailed grouse is designated as an ‘upland game bird’ under the provincial *Wildlife Act*, meaning that hunting and collection of this species is controlled under the *Act*. The Committee on the Status of Wildlife in Canada (COSEWIC) has not completed a formal assessment or determined a rank for the sharp-tailed grouse nationally (COSEWIC 2016). Consequently, the species has not been listed on the federal *Species at Risk Act*. Internationally, the species has been designated as a species of ‘Least Concern’ by the International Union for Conservation of Nature and Natural Resources (IUCN) (BirdLife International 2012).

Although common and widespread in Alberta, sharp-tailed grouse populations appear to be declining (GoA 2016). By the mid 1990’s, population trend data for sharp-tailed grouse in Alberta indicated population declines ranging from 50% to 70% in some areas over the last 30 years (Goddard 1995, Jones and Lee 2000). Increased agricultural production, the loss of native prairie habitat and intensive livestock grazing have contributed to a decline in sharp-tailed grouse populations in Alberta and across their range in North America (Millar 1999, Giesen and Kobriger 1997, GoA 2016).

1.2 Ecology

During the spring breeding season, male sharp-tailed grouse gather and perform ritual courtship displays on “leks” or “dancing grounds”. Male sharp-tailed grouse gather on leks as early as late February (depending on geographic location), with peak activity for female lek visitation, highest male visitation and greatest activity on leks occurring in April (Pepper 1972, Alberta

Conservation Association [ACA] 2010, GoA 2013a). Breeding displays typically occur for about 2.5 to 3.5 hours in the morning, beginning 30 to 40 minutes before sunrise, and continue again for about 1.5 hours before sunset (Pepper 1972).

Female sharp-tailed grouse begin to construct a nest scrape at approximately the same time or before visiting dancing grounds in the spring (Millar 1999). The timing of nesting and breeding is affected by snowfall, with earlier nesting dates recorded in years of little snow in March and April, and later nesting dates in April and May when snow persists (Bergerud and Gratson 1988, ACA 2010). Sharp-tailed grouse have an average clutch of 12 eggs (Amman 1957). Eggs are laid daily and the 21 to 24 day incubation period begins after the last egg is laid (Millar 1999, ACA 2010). Hatching occurs from the first of June to the middle of July, with peak hatching occurring during approximately the second or third week of June (Pepper 1972). Sharp-tailed grouse chicks are precocial and leave the nest shortly after hatching in search of brood rearing habitat. Broods may remain in the vicinity of nests if suitable brood-rearing habitat is present or they may move longer distances to brood-rearing habitat (Goddard and Dawson 2009). Chicks are capable of flying to some degree by 10 days of age, and disperse from the brood when they become fully independent at six to eight weeks (Millar 1999).

Sharp-tailed grouse mortality is highest for chicks within their first 0 to 14 days of life (Goddard and Dawson 2009). In a study in northeastern British Columbia, only 34% of chicks survived to 35 days of age (Goddard and Dawson 2009). Other literature has noted higher (50%) survival rates (ACA 2010). Mortality in chicks is primarily attributed to predation from mammals and other avian species. Weather and time of hatching have also been noted as important variables for determining brood success (Goddard and Dawson 2009). In addition, it was proposed that broods that moved from nesting areas within the first few weeks following hatching were more susceptible to mortality.

1.2.1 Diet

Sharp-tailed grouse are omnivorous and consume fruits, green leaves, buds and insects (Millar 1999). Food is not considered limiting to sharp-tailed grouse populations or breeding densities (Bergerud 1988). The vegetation used by sharp-tailed grouse for cover is typically also used for food, with cover value often selected over forage value (Bergerud 1988, Millar 1999). Table III-3 provides a brief description of typical sharp-tailed grouse diets at different life stages and seasons of the year.

Table III-3 Sharp-tailed Grouse Diet

Grouse Life Stage and Season of Year	Dietary Description
Chick Diet	• Insects, including grasshoppers, beetles and ants, comprise the dominant diet of chicks for their first two weeks of life. • At 12 weeks of age, chick diets resemble adult diets, consisting of approximately 90% plant material (Millar 1999).
Adult Spring Diet	• Favoured spring forage includes aspen (*Populus tremuloides*) catkins, new forb growth such as dandelions (*Taraxacum* spp.) and remaining available fruits from the previous season (Millar 1999).

Adult Summer Diet	• In the Nebraska sandhills, summer sharp-tailed grouse diets consisted of 91% plant material and 5% insects (Kobriger 1965). • Important summer forages include clover (*Trifolium* spp.), rose (*Rosa* spp.) and dandelion.
Adult Fall and Winter Diet	• Woody plant fruits, seeds, buds and leaves from hawthorn (*Crataegus* spp.), rose, buckbrush (*Symphoricarpos occidentalis*), choke cherry (*Prunus virginiana*), saskatoon (*Amelanchier alnifolia*), currant (*Ribes* spp.), Russian olive (*Elaeagnus angustifolia*), creeping juniper (*Juniperus horizontalis*), cottonwood (*Populus* spp.) and aspen constitute the majority of fall and winter diets (Millar 1999). • Where waste grains from croplands are less than 50 m from woody cover, grain is also consumed during the winter (Millar 1999).

1.2.2 Predators

Sharp-tailed grouse and their nests are vulnerable to predation from numerous types of raptors, corvids (crows, jays, magpies and ravens) and mammalian predators such as red foxes (*Vulpes vulpes*), coyotes (*Canis latrans*), striped skunks (*Mephitis mephitis*) and weasels (*Mustela* spp.) (Millar 1999).

1.3 Habitat Requirements

1.3.1 General

A mosaic of plant communities, including native grassland and shrub mixtures with extensive ecotone habitat, are considered important habitat components for maintaining sharp-tailed grouse populations (Pepper 1972, Moyles 1981, Swenson 1985).

As part of the Multiple Species at Risk (MULTISAR) project, a Habitat Suitability Index (HSI) model was developed for sharp-tailed habitat use in the Milk River Watershed (Jones 2004a). Native prairie cover and percent shrub cover were the two variables included in this HSI model. Areas with 75% to 100% native prairie cover and between 5% to 15% shrub cover were rated as highly suitable habitat for sharp-tailed grouse. Native prairie represents nesting, hiding and brood-rearing habitat, while shrub cover represents a component of nesting and winter habitat. Based on this model, the two largest areas with the greatest habitat potential for sharp-tailed grouse in the Milk River Watershed are the Milk River Ridge area in the west and the Sage Creek area in the southeast (Jones 2004). A second HSI model was developed for sharp-tailed grouse habitat use in the headwaters of the Oldman River Watershed (Jones 2004b). Similar variables and criteria were used as the aforementioned HSI model, with the addition of <5% cover for trees, and similar results were obtained.

1.3.2 Lek (Dancing Ground) Site

Sharp-tailed grouse leks or dancing grounds are commonly characterized by low, sparse vegetation that allows for good visibility and uninhibited movements (Millar 1999). Leks are most often located on a slight rise, such as a low ridge or knoll or in open flat areas, that provide “wide-viewing horizons” in all directions (Baydack 1988, Millar 1999). The stability and availability of surrounding suitable nesting habitat is also an important factor in the establishment and consistency of use of leks (Bergerud and Gratson 1988). Availability of food, roosting, loafing and escape cover in the vicinity are other determinants of lek location and occupancy (Pepper 1972, Millar 1999).

1.3.3 Nesting Habitat

Roersma (2001) found that sharp-tailed grouse nests were located an average distance of 1.1 km from their associated leks in the Milk River Ridge area. Preferred sharp-tailed grouse nesting habitat includes lush and dense residual growth of grasses and sedges in association with short shrubs such as rose and buckbrush (Pepper 1972, Millar 1999, Roersma 2001). Treed bluffs are not typically used as nesting habitat (Millar 1999). Vegetation heights greater than 24.5 cm but less than 6 m are used for nesting (Christenson 1970, Pepper 1972). A minimum cover height of 30.5 cm, a high percentage of overhead cover (75%) and a minimum of 15 cm have been noted as spring sharp-tailed grouse nesting requirements (Amman 1957, Christenson 1970, Kobriger 1980). Known nesting sites for sharp-tailed grouse in northeastern British Columbia included 32% shrub, 38% grass and 27% forb cover where overhead cover was 57% (Goddard *et al*. 2009). Roersma (2001) noted that sharp-tailed grouse nests on the Milk River Ridge contained more woody (shrub) cover and less graminoid cover than random sites. More research is needed to assess the effect of litter cover thresholds and sharp-tailed grouse nesting success. Thresholds of less than 25% litter cover and greater than 25% grass cover have been identified as predictors of increased nest success in greater prairie chickens in Missouri (McKee *et al.* 1998).

1.3.4 Brood-rearing Habitat

Brood-rearing habitat is selected to ensure predator avoidance and adequate food availability (Goddard *et al.* 2009). Sharp-tailed grouse chicks require a high-protein diet of primarily insects during the first weeks of their lives (Bergerud and Gratson 1988). During this critical period chicks cannot thermoregulate and must find suitable foraging habitat within close proximity of the nest site (Bergerud and Gratson 1988). To accommodate the high protein dietary requirements of chicks, broods use a greater variety of plant cover types than nesting hens and prefer warm, more open areas with a higher percentage of forbs and insects (Pepper 1972, Bergerud 1988). During the day, broods typically use less dense cover in the early morning and evening but seek areas of taller vegetation and more dense grass or woody cover at midday (Pepper 1972, Christenson 1970, Moyles 1981). Broods benefit from a mosaic of cover types including non-use grasslands, edges of heavily grazed pastures and woody draws (Christenson 1970, Roersma 2001, Goddard *et al.* 2009) noted that brood rearing sites in the Milk River Ridge area had greater grass cover and reduced litter cover than random sites.

1.3.5 Roosting and Winter Cover

When snow conditions permit, sharp-tailed grouse will tunnel into snowdrifts to shelter from strong winds and severe winter temperatures (Millar 1999, ACA 2010). Woody cover including aspen bluffs and shrubby or riparian hardwood draws also provide essential thermal cover for sharp-tailed grouse during cold winters as well as concealment from predators (Swenson 1985, Giesen and Connelly 1993).

1.3.6 Area Requirements

The minimum amount of contiguous suitable habitat that is required before an area will be occupied by a sharp-tailed grouse population constitutes the “minimum habitat area” (Prose 1987). The minimum habitat area required by sharp-tailed grouse populations will inevitably vary according to the quality and structure of available vegetation cover as well as with varying

predator densities (Bergerud 1988). Kobriger (1980) calculated the minimum habitat area for sharp-tailed grouse in North Dakota to be a 5.3 km^2 circle based on a radius of 1.3 km, with the lek considered the central point of activity. The radius of 1.3 km was defined according to the mean distance from known leks to nest sites. In the aspen parkland of Saskatchewan, Pepper (1972) determined that a minimum of 0.24 km^2 of suitable grassy and/or shrubby vegetation around a lek was needed to attract an average of six male sharp-tailed grouse. However, to consistently attract a viable male population to leks each spring, Pepper (1972) proposed that between 1.6 km^2 and 3.2 km^2 of suitable surrounding vegetation was necessary. Roersma (2001) calculated the average total nesting area for five sharp-tailed grouse leks in the Milk River Ridge area to be 1.48 km^2.

1.4 Grazing and Sharp-tailed Grouse

Livestock grazing has the potential to impact sharp-tailed grouse due to removal of vegetation cover, alteration of plant species composition and potential trampling of eggs and nests. Changes in vegetation cover and plant species composition can have beneficial or detrimental impacts to the availability of plant or insect food sources and the quality of sharp-tailed grouse nesting, foraging, escape or wintering cover. In general, heavy grazing for prolonged periods can have a negative impact on critical sharp-tailed grouse breeding and wintering habitats (Pepper 1972, Nielsen 1981, Giesen and Connelly 1993). Light to moderate grazing is considered more appropriate for sustaining the quality of sharp-tailed grouse habitats (Nielsen 1981). Light to moderate stocking rates allow for the retention of residual grass cover and shrub patches that provide important nesting and escape cover. McNew *et al.* (2015) suggest that grazing management systems should resemble historical fire and grazing regimes, which were patchy in distribution. This type of management system, patch-fire grazing, appeared to improve greater prairie-chicken (*Tympanuchus cupido*) performance compared to annual burning and early stocking. It is especially important to minimize the impacts of livestock to leks and surrounding nesting cover during the breeding season as well as to minimize impact to riparian areas (Giesen and Connelly 1993).

1.5 Grazing Systems and Sharp-tailed Grouse Habitat Management

Several studies have examined the effects of various grazing systems on the habitat needs and reproductive success of sharp-tailed grouse (Kobriger 1980, Mattise *et al.* 1981, Nielsen 1981, Grosz 1985, Sedivec *et al.* 1990, Kirby and Grosz 1995). The results of these studies are summarized in Table III-4. Overall, no one grazing system appears to be unanimously preferred. Stocking rate and the timing and distribution of cattle are considered decisive factors influencing the potential benefits or detriments of each grazing system.

Table III-4 provides an overview of eight grazing systems and their potential positive and negative effects on sharp-tailed grouse and their habitat. A grazing system is a tool used to control the spatial distribution, timing, intensity and frequency of livestock grazing (Holechek *et al.* 2003). Applied research is needed to properly assess the effects of various grazing systems on sharp-tailed grouse in the Milk River and South Saskatchewan River watersheds.

Table III-4 Grazing Systems and Sharp-tailed Grouse Habitat Management

Grazing System	Discussion
Continuous (Season-long) Grazing	
Advantages:	Under moderate stocking rates, continuous grazing often leads to patch grazing effects in fescue prairie and to a lesser extent in mixedgrass prairie, with areas that are consistently re-used and areas that receive little use. Patch grazing is typically the result of continuous grazing where forage supply exceeds livestock demand (Spedding 1971). Variably grazed patches help to create a heterogeneous habitat mosaic. Ungrazed or less used patches offer suitable nesting, roosting and escape cover. Grazed patches will typically stimulate increaser forb growth and may offer microsites for insect production. Studies have shown that several grasshopper species are more numerous in heavily grazed areas and prefer ranges with sparse grass stands and a high forb component (Holmes *et al.* 1979, Holechek *et al.* 2003). The shorter vegetation of grazed patches near to taller, thicker cover is ideal foraging habitat for sharp-tailed grouse chicks that may otherwise be inhibited by dense vegetation or thick litter. Roersma (2001) noted that brood-rearing sites had less litter cover than random sites in the Milk River Ridge area. Kobriger (1980) recommended season-long grazing at moderate stocking rates that allow a 50% utilization rate as a strategy to prevent uniform grazing and promote vegetation heterogeneity for sharp-tailed grouse in mixedgrass prairie in North Dakota. Nesting hens in this study demonstrated a preference for grassy upland habitats. When quantity and quality of surrounding grasslands was low, sharp-tailed grouse were observed to nest in shrubby lowland draws. Mattise *et al.* (1981) noted that the average height and density of vegetation was significantly greater in their season-long gazing system compared to their deferred-rotation system. In this study, conducted in the mixedgrass prairie of North Dakota, grazing rates averaged 2.3 ac per animal unit month, and pasture sizes averaged approximately 605 ac (Mattise *et al.*1981).
Disadvantages:	Continuous grazing under intensive stocking rates can have obvious detrimental impacts to sharp-tailed grouse habitat by creating uniformly heavily grazed conditions. Lack of adequate carry-over and uniform grazing decreases nesting, foraging and escape cover for sharp-tailed grouse. In addition, riparian areas are often heavily impacted by persistent cattle use under continuous grazing systems, particularly if alternate water supplies or fencing are not in place. Persistent use of riparian areas can eliminate deciduous trees or shrubby vegetation that form an important part of sharp-tailed grouse winter habitat.

Rotational Grazing - Deferred Rotation	
Advantages:	Deferred spring grazing has the obvious advantage of minimizing possible grazing or trampling disturbance to nesting or breeding sharp-tailed grouse during a critical time of year. Deferring grazing early in the season, particularly in fescue prairie, would have the added advantage of improved plant vigour and sustained productivity (Willms and Fraser 1992). Pepper (1972) suggests that the carrying capacity for sharp-tailed grouse is limited by the availability of large acreages of ungrazed grass-shrub and hayland within 1.6 km of a lek. Deferred spring grazing would therefore provide sharp-tailed grouse with ungrazed or lightly grazed areas within proximity to the lek during the nesting season (Millar 1999). Deferred spring grazing was found to be mutually beneficial to livestock, waterfowl and sharp-tailed grouse production in a study conducted at the Central Grasslands Research Center in south -central North Dakota (Kirby and Grosz 1995).
Disadvantages:	The potential advantages of deferred grazing for enhancing sharp-tailed grouse habitat depend on the period of deferment, the intensity of grazing and the degree of carry-over that remains after the grazing season. Grazing later in the season could affect the amount of cover available to sharp-tailed grouse early the next season if grazing is intense and residual carry-over material is not retained.
Complementary Grazing	
Advantages:	As with deferred grazing systems, complementary grazing allows for undisturbed residual nesting cover and limits disturbance to sharp-tailed grouse during nesting and brood-rearing. Deferred grazing pressure early in the season, as mentioned, can improve the health, productivity and sustainability of fescue grasslands in particular. Based on the results of a six-year study, Prescott and Wagner (1996) concluded that complementary grazing in combination with rotational grazing improved range condition (health), grass yields and litter reserves in native prairie. These improvements were noted to provide a mosaic of habitats suitable for a wide spectrum of upland nesting birds.
Disadvantages:	Complementary grazing is particularly beneficial if existing cropland or hayland can be converted to permanent cover (*i.e.*, tame pasture). However, if a complementary grazing system is established by converting native prairie, the benefits of this system may be outweighed by the loss of higher quality native habitat (Pepper 1972, Prescott and Wagner 1996). The value of tame pasture as sharp-tailed grouse habitat is dependent on the size of the pastures in relation to surrounding native prairie and the amount of woody vegetation that is retained (Pepper 1972, Moyles 1981, Swenson 1985).

Season-of-Use Grazing	
Advantages:	Season-of-use grazing has good potential to enhance sharp-tailed grouse habitat for ranching operations that encompass mixedgrass and fescue prairie. The best season-of-use for mixedgrass prairie is in late spring and early summer, while fescue grassland benefits from later season grazing in late summer, early fall or winter. This type of grazing operation would likely result in improved range health and would have the potential to provide good quality undisturbed residual nesting cover early in the season (Pepper 1972, Giesen and Connelly 1993, Millar 1999).
Disadvantages:	As with all other grazing systems, the stocking rate and percentage of vegetation utilization under season-of-use grazing will ultimately influence its benefit to both range health improvement and sharp-tailed grouse habitat enhancement.
Rotational Grazing (General)	
Advantages:	Rotational grazing systems, including switchback, deferred-rotation and rest-rotation grazing, allow for timed sequences of grazing and rest periods in smaller-sized fields (Holechek *et al.* 2003). Rotational grazing reduces selective grazing by encouraging use of plant groups differing in their season of growth and allows for seed production, seedling establishment and restored plant vigour (Adams *et al.* 1991). Enforcing periods of rest allows undisturbed areas with residual vegetation to be retained and thereby promotes nesting habitat for sharp-tailed grouse. Periods of rest are also beneficial to minimizing disturbance to sharp-tailed grouse during critical periods. Rest periods and periodic deferred early-season grazing can also promote the recovery of riparian plant communities that would otherwise receive consistent heavy grazing pressure. Riparian areas are important foraging, winter and escape cover for sharp-tailed grouse.
Disadvantages:	A noted disadvantage of rotational grazing is the potential creation of uniform grazing effects due to improved cattle distribution. Uniform grazing effects are accentuated when high stocking densities are used, forcing cattle to use the entire area available for grazing. As sharp-tailed grouse favour a mosaic of vegetation structure to provide them with suitable shelter, nesting and foraging habitats, uniform grazing effects may be detrimental to their productivity and survival. The creation of uniform grazing effects can be controlled if stocking rates are reduced.

Rotational Grazing - Deferred and Rest-Rotation	
Advantages:	Deferred-rotational grazing was recommended over other types of grazing systems as an effective means of providing upland bird nesting cover while also optimizing beef production per acre in two comparative studies in the mixedgrass prairie of North Dakota (Sedivec *et al.* 1990). Sedivec *et al.* (1990) found the greatest proportion of sharp-tailed grouse nests (51%) in the twice-over deferred-rotation grazing system. (Twice-over grazing involves two rotations through a field in one season). All nests that were found during this study were initiated before the grazing season began (fourth week in May) or were in ungrazed pastures during rotations (Sedivec *et al.* 1990). Deferred-rotation fields were grazed at a stocking rate of 2.7 AUM/ha in this study. Sedivec *et al.* (1990) recommended deferred-rotational grazing as a means of promoting the maximum amount of undisturbed cover available for nesting upland birds until early July. In a similar study, Grosz (1985) also recommended twice-over deferred-rotational grazing to promote nesting cover for sharp-tailed grouse. Twice-over deferred-rotation fields promoted nesting habitat as they had 60% of the vegetation in Visual Obstruction Readings (VOR's) of 15 cm or greater (Grosz 1985). Grosz (1985) found 70% of the nests in his study were located in vegetation with a VOR of 15 cm or greater.
Disadvantages:	Kobriger (1980) suggests that deferred-rotational grazing contributes to uniform grazing effects as cattle are often grazed beyond the 50% utilization point. This occurs when movement between fields is based on calendar dates and not vegetation utilization. Similarly, Mattise *et al.* (1981) caution that deferred rotation grazing may intensify use and reduce cover in areas of a field that would normally only be lightly grazed under season-long grazing. Nielsen (1981) noted that at high stocking rates, rest-rotational grazing can concentrate grazing effects and reduce the cover value of woody draws and reduce nesting cover close to leks. Nielsen noted that despite heavy grazing pressure, sharp-tailed grouse did not move from their traditional use areas. Nielsen concluded that despite the benefits of better quality habitat in the rest field, these benefits may not exceed the harmful effects of the intensive grazing on the other three fields as sharp-tailed grouse did not adjust their use areas in relation to changing grazing pressures.
Rotational Grazing – High Intensity Grazing	
Advantages:	Seasonal intensive grazing can be beneficial if it occurs in an area of low quality or low use sharp-tailed grouse habitat (such as tame pastures), and it is used to defer spring use of other native fields with lek sites.

Disadvantages:	The negative impacts of overgrazing and intensive grazing by cattle on sharp-tailed grouse habitat are well-documented (Sisson 1970, Pepper 1972, Nielsen 1981, Giesen and Connelly 1993). Intensive livestock grazing for prolonged periods or early in the season is harmful to sharp-tailed grouse habitat by reducing necessary cover for nesting, shelter, roosting and winter foraging. Excessive trampling, grazing and browsing near to leks and riparian areas are especially detrimental to sharp-tailed grouse habitat. High stocking rates are particularly damaging to fescue grasslands.
Riparian Area Grazing	
Advantages:	Grazing systems that are designed to control and minimize livestock impact of riparian areas will have benefits to improving sharp-tailed grouse habitat. Reduced pressure on riparian areas with forests and shrubs will provide higher quality fall and winter cover and foraging habitat for sharp-tailed grouse. Pepper (1972), Moyles (1981), Nielsen (1981), Swenson (1985) and Giesen and Connelly (1993) all comment on the importance of protecting riparian areas from excessive overgrazing by livestock due to the importance of riparian shrubs and trees as sharp-tailed grouse habitat. Grazing systems that benefit riparian health limit cattle use of riparian areas during their most vulnerable periods, including spring when streambanks are soft, and autumn when riparian vegetation is more palatable than upland vegetation.
Disadvantages:	The benefits of riparian grazing systems depend on suitable fence and alternate water source placement to avoid concentrating cattle use close to leks or in suitable nesting habitat early in the season.

1.6 Beneficial Management Practice Recommendations

The following beneficial management practices are recommended to protect important sharp-tailed grouse habitat components in the Milk River and South Saskatchewan River watersheds and to prevent disturbance to sharp-tailed grouse during critical periods. These recommendations should be reviewed and amended as additional knowledge becomes available. A “breeding complex” encompassing all land within a 2 km radius of a lek was recommended as the management unit for Columbian sharp-tailed grouse (*Tympanuchus phasianellus columbianus*) (Giesen and Connelly 1993). For the Milk River Ridge, Roersma (2001) suggests a management unit referred to as the “total nesting area”, an area that encompasses all nests associated with a lek. Roersma (2001) calculated the average total nesting area for five leks to be 1.48 km^2.

1.6.1 General Recommendations

- Conserve remaining native prairie from cultivation and development.
- Limit disturbance within a minimum 1.48 km^2 total nesting area around leks during the breeding and nesting seasons (*i.e.*, March to June) (Baydack1987, Giesen and Connelly 1993, Roersma 2001). Sharp-tailed grouse can be displaced from leks due to human disturbance (Baydack 1987). Disturbance to leks during the breeding and nesting season can, therefore, negatively impact sharp-tailed grouse reproductive success.

- Abide by set-back distances and timing restrictions outlined by Alberta Environment and Parks (AEP) for human activities, including industrial development, near sharp-tailed grouse leks (GoA 2011, 2013a). From March 15th to June 15th, AEP recommends a 500 m set-back from an active lek for low, medium and high impact activities such as wellsites, powerlines and pipelines. From June 16th to March 14th, AEP recommends a 100 m set-back for low and medium impact activities and 500 m set-back for high impact activities (GoA 2013b).
- As per the Enhanced Approval Process (EAP), install perch preventers on above-ground structures within 1,000 m of a lek to reduce depredation rates (GoA 2013b).
- As per the EAP, use noise reduction equipment to muffle operational noise within 500 m of a lek (GoA 2013b).
- Attempt to avoid surface facility densities in excess of 1 well pad/2.5 km^2 (GoA 2013b).
- Conduct pre-development wildlife surveys to locate sharp-tailed grouse leks in order to plan developments in accordance with EAP and Sensitive Species Inventory Guidelines (GoA 2013a,b). Surveys need only be conducted in areas with suitable sharp-tailed grouse habitat. All survey data should be submitted to AEP for inclusion in the Fish and Wildlife Management Information System.
- Practice zero tillage of croplands and retain stubble fields within 1 km of woody draws or breaks to provide sharp-tailed grouse with an alternate winter food supply (Goddard 1995).
- Use flushing bars and delay mowing of hayland within 1.6 km of active leks until mid-July to allow nesting hens with broods to fledge successfully (Goddard 1995).
- Promote organic farming practices and minimize the use of pesticides within 1.6 km of active leks. Pesticides can have direct impacts on sharp-tailed grouse survival and reproduction and can result in reduced insect and plant food sources (Millar 1999).
- Retain shrubby and woody draw vegetation mosaics across fields and within the total nesting area (Moyles 1981, Roersma 2001).
- Reclaim disturbed native vegetation where possible and revegetate heavily impacted riparian corridors or woody draws.
- Manage nesting and brood-rearing habitats together to decrease likelihood of chick and brood mortality (Goddard and Dawson 2009).

1.6.2 Grazing Recommendations

Final decisions regarding the type of grazing system and calculation of suitable stocking rates that would best benefit sharp-tailed grouse need to be made on a case-by-case basis. Local conditions need to be considered, including vegetation type, range and riparian health, distribution of valued habitat components and dispersion of lek sites across the landscape. No one grazing system can be universally applied, and each must be tailored to local environmental conditions (Guthery 1995). Control and flexibility over stocking rates and dispersion of livestock use are two key properties of an optimum grazing system (Guthery 1995). Appropriate grazing systems should be developed site-specifically for participating ranches MULTISAR conservation program.

Grazing systems designed to enhance sharp-tailed grouse habitat should recognize the importance of native grass and shrubs as key habitat components for maintaining viable sharp-tailed grouse populations. In addition, monitoring and managing lek sites and surrounding nesting habitat should be a priority.

The following grazing management principles should be applied to maintain habitat for sharp-tailed grouse:

- Limit disturbance within a minimum 1.48 km^2 total nesting area around leks during the breeding and nesting season (*i.e.*, March to June) (Baydack1987, Giesen and Connelly 1993, Roersma 2001).
- Defer grazing during the spring breeding and nest initiation periods (*i.e.*, March to late May) (Sedivec *et al.* 1990, Kirby and Grosz 1995). Where possible, a grazing commencement date of mid-June is recommended to avoid grazing disturbance during breeding, nesting and peak hatching (Pepper 1972).
- Provide sufficient residual grass and shrub cover within the breeding complex (Roersma 2001, Giesen and Connelly 1993, Millar 1999).
- Use appropriate stocking rates to ensure sufficient carry-over and maintained plant vigour and range condition (Pepper 1972, Kobriger 1980, Mattise *et al.* 1981).
- Make seasonal adjustments in stocking rates based on fluctuations in precipitation and time of use.
- Avoid intensive, high stocking rate grazing systems and discourage uniform utilization of fields (Pepper 1972, Sisson 1975, Kobriger 1980, Mattise *et al.* 1981).
- Create heterogeneous habitat with areas of varying species diversity and a gradation of short to tall and dense to light residual vegetation and litter cover (Sisson 1975, Kobriger 1980, McNew *et al.* 2015).
- Encourage periods of rest and defer early-season grazing in fescue prairie to improve range health (Grosz 1985, Sedivec *et al.* 1990).
- Base rotational grazing systems on percentage of vegetation utilization and not scheduled calendar dates (Kobriger 1980).
- Convert cropland to tame pasture and implement complementary grazing where practical (Prescott and Wagner 1996).
- Manage livestock grazing in riparian areas to protect riparian shrubs and deciduous trees (Pepper 1972, Moyles 1981, Nielsen 1981, Swenson 1985 and Giesen and Connelly 1993).
- Fence out or minimize livestock use in woody vegetation considered to be critical sharp-tailed grouse wintering habitat (Goddard 1995).
- Develop strategic salting locations away from leks, stock water and woody draws. Continuously move salt throughout the grazing season. Avoid placing salt or water facilities within a 1.48 km^2 total area around leks during the breeding and nesting season (*i.e.*, March to June).
- Monitor and maintain records of active leks, to range health, and yearly grazing records, including livestock numbers, class, breed, take-in and take-out dates.

1.7 Research Recommendations

Due to its importance as an upland game bird, the ecology, habitat requirements and effects of grazing on sharp-tailed grouse have been fairly well-studied. Most recently, Roersma (2001) examined the nesting and brood-rearing ecology of plains sharp-tailed grouse in the Milk River Ridge area in Alberta. Further studies could be done to assess the effect of litter cover thresholds on sharp-tailed grouse nesting success.

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C. BURROWING OWL

1 INTRODUCTION

1.1 Background

The purpose of this report is to summarize the ecology and habitat requirements of the burrowing owl (*Athene cunicularia*) in southern Alberta. Based on this information and supporting scientific studies, various grazing systems are compared in terms of their potential implications to burrowing owl ecology and habitat. This discussion is followed by a summary of recommended beneficial management practices (BMPs) to enhance burrowing owl habitat in the Milk River and South Saskatchewan River watersheds in Alberta, with broader application to the range of this species within the Grassland Natural Region of Alberta (Natural Regions Committee [NRC] 2006). Lastly, a brief summary of additional research needs that would improve our understanding of burrowing owl ecology, habitat requirements and response to land management is presented.

The burrowing owl is one of 10 owl species in Alberta. Burrowing owls are small with light brown colouration, brown spots, a rounded head with no ear tufts, long, lightly feathered legs and a stubby tail (Semenchuk 1992). In Alberta, the species occurs only in the Grassland Natural Region.

The burrowing owl is listed as 'At Risk' of extirpation in Alberta under the General Status listing (Government of Albert [GoA] 2016). The species has been designated as Threatened under provincial *Wildlife Act*. Nationally, the burrowing owl was assessed by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) in 1979 and designated as 'Threatened' (Wedgwood 1979). In 1995, a status re-examination led the species to be up-listed to 'Endangered' status in Canada, a status the species currently retains (Wellicome and Haug 1995, COSEWIC 2006, 2016). In 2003, the burrowing owl was listed as 'Endangered' under Schedule 1 of the *Species at Risk Act* (*SARA*). Internationally, the International Union for Conservation of Nature and Natural Resources (IUCN) has designated the burrowing owl a species of 'Least Concern' (BirdLife International 2012).

The burrowing owl's historic range covered roughly 450,000 km^2 across western Canada, including British Columbia, Alberta, Saskatchewan and Manitoba (Environment Canada 2012), By the mid-2000s, the species' range was only 36% the size of its historic extent *(i.e.*, 160,000 km^2) (Environment Canada 2012). The population of burrowing owl declined by as much 90% in 1990s, becoming extirpated in British Columbia and Manitoba (Environment Canada 2012). The decline slowed between the mid-1990s and mid-2000s, but remained above 50% (Environment Canada 2012). The 2004 burrowing owl population in Alberta was estimated at 200 to 400 breeding pairs and declining dramatically across Canada (GoA 2016).

Agricultural destruction of native prairie habitat, intensified land-use, pesticide use, eradication efforts to control burrowing rodents and an increase in prairie predator populations are contributing factors to the continuing population decline of the burrowing owl in Canada (Hjertaas *et al.* 1995, Dechant *et al*. 2002; GoA 2016), although the exact cause of the decline is

unknown (COSEWIC 2006). Confounding the issue of the species' decline is the fact that seemingly suitable habitat exists in areas where the species has disappeared (Environment Canada 2012).

A national Recovery Strategy for the burrowing owl was released by the federal government in 2012 (Environment Canada 2012). A Recovery Plan for the burrowing owl was released by the provincial government that same year (Alberta Environment and Sustainable Resource Development 2012).

1.2 Ecology

Burrowing owls overwinter in Mexico and Texas and typically return to their breeding grounds in Alberta between early April and early May (Wellicome 1997). Arrival dates vary with the severity of spring weather conditions (Wellicome 1997). Egg-laying begins between late April and late May in Alberta, with eggs incubated for approximately four weeks (Hjertaas *et al*. 1995, Wellicome 1997). Clutch sizes range from six to 11 eggs (Wellicome 1997). Hatchlings are altricial and rely on being fed by their parents during their first few weeks of development. Chicks become fully independent at 60 to 70 days old (Wellicome 1997). Burrowing owl nests are usually lined with dried, shredded cow or horse manure, possibly to mask nest odours to avoid predation (Wellicome 1997, Dechant *et al.* 2002).

1.2.1 Diet

Burrowing owls are generalist predators and primarily consume small mammals and arthropods (Wellicome 1997, Dechant *et al.* 2002). Food intake (prey abundance) was found to be more limiting during brood-rearing than during egg-laying in an experimental study conducted in Saskatchewan (Wellicome *et al.* 1997). Burrowing owls that received supplemental food during the nestling stage produced 41% more fledglings than burrowing owls that were not given supplemental food (Wellicome *et al.* 1997). In Alberta, deer mice (*Peromyscus maniculatus*) and meadow voles (*Microtus pennsylvanicus*) made up as much as 90% of burrowing owl diets by weight early in the breeding season (Schmutz *et al.* 1991, Haug *et al.* 1993). In south-central Montana, the breeding season diet of burrowing owls consisted of 72% small mammals, primarily prairie voles (*Microtus ochrogaster*) and secondarily mice (*Peromyscus* spp.). In this study, insects were difficult to quantify from pellets and were likely under-represented in the sample. In Alberta, dung and carrion beetles are often consumed early in the breeding season, and in some years, grasshoppers form a significant part of burrowing owl diets later in the season, constituting up to 35% of the mass consumed (Wellicome 1997). In Saskatchewan, invertebrates comprised 93% of the prey species consumed by burrowing owls (Haug 1985).

1.2.2 Predators

Burrowing owls have numerous potential predators including striped skunks (*Mephitis mephitis*), American badgers (*Taxidea taxus*), coyotes (*Canis latrans*) and various raptor species (Wellicome 1997).

1.3 Habitat Requirements

1.3.1 General

Burrowing owls are most abundant in the Mixedgrass and Dry Mixedgrass Natural Subregions of Alberta (Alberta Sustainable Resource development [ASRD] and Alberta Conservation Association [ACA] 2005, NRC 2006), and seldom occur in fescue grasslands (Wellicome 1997). Burrowing owls do not dig their own burrows but instead rely on burrowing mammals such as Richardson's ground squirrels (*Spermophilus richardsonii*), black-tailed prairie dogs (*Cynomys ludovicianus*) and American badgers to excavate their nest sites (Dechant *et al.* 2002). Badger hole-sized burrows are preferred (Poulin *et al.* 2005). In Canada, black-tailed prairie dogs only occur in and near the Frenchman River Valley in southern Saskatchewan (COSEWIC 2011). Black tailed-prairie dogs are more common in the arid short and mixedgrass prairies of the Great Plains of the United States. The largest remnant populations of this rodent are currently found in South Dakota, Wyoming, Montana and Mexico (U.S. Fish and Wildlife Service 2003). Burrows are an essential feature of burrowing owl habitat as they provide shelter from predators, protection of eggs and small young as well as shelter from adverse weather (Clayton and Schmutz 1999, Poulin *et al.* 2005).

As part of the Multiple Species at Risk (MULTISAR) conservation program in southern Alberta, a Habitat Suitability Index (HSI) model was developed for burrowing owl habitat use in the Milk River Watershed (Skiftun 2004). Using information on known locations as well as the biology and ecology of wildlife species, HSI models use various physical and habitat variables to predict where suitable habitat for the species may be found on the landscape. The HSI model developed for the burrowing owl included four variables: native prairie coverage; soil texture; shrub/tree coverage; and distance from linear disturbances (*i.e.*, roads). Native prairie cover is the most important factor in the formula and is weighted more strongly than the other variables. According to the HSI model, native prairie with moderate to moderately coarse textured soils (silty loam to sandy loam), 0% cover of trees and shrubs and that is situated greater than 800 m from linear disturbances represents ideal burrowing owl habitat.

Also as part of the MULTISAR program, a Resource Selection Function (RSF) model was developed in 2014 for burrowing owl habitat use in southern Alberta (Gan *et al.* 2014). RSFs are similar to HSI models but are thought to be more robust due to their rigorous and empirical evaluation of model performance using statistics (Boyce *et al.* 2002). RSFs estimate relative probability of occurrence for species of concern on the landscape based on various physical and anthropogenic variables. As part of the MULTISAR RSF development, second order (home range within the species range) and third order models (site-specific habitat within home range) were developed (Gan *et al.* 2014). The best-fit second order model predicting optimal burrowing owl home ranges included five variables: proportion of favourable soil types; percent grass cover; percent water cover; percent shrub cover; and road density. According to the second order model, relative probability of use increases as grass cover increases, in areas with lower amounts of water, favourable soil types and higher road densities, and use decreases as shrub cover increases. The best-fit third order model predicting burrow locations on the landscape included seven variables: distance to grass (linear and quadratic terms); slope at location; presence of favourable soil; distance to water; and presence of natural land. Relative probability of burrows on the landscape was determined to be a non-linear function of distance to grass. Flat terrain and

native prairie were both preferred habitat characteristics. Soil type is important as it affects burrow structural integrity. Proximity to water was also important, likely due to prey increased availability near water (Gan *et al*. 2014).

1.3.2 Nesting Habitat

Burrowing owl nesting habitat is characterized by available nest burrows, short or sparse vegetation and open, treeless plains (Wellicome 1997; Zarn 1974). Pastures grazed by livestock provide the majority of nesting habitat for this species in Canada (ASRD and ACA 2005; Wellicome 1997). Short vegetation at nest burrows is thought to be important to allow burrowing owls to easily detect predators (Felskie pers. comm.). Clayton and Schmutz (1999) reported that burrowing owls near Hanna, Alberta preferred shorter (≤ 10 cm) grasses for both nesting and roosting and that all nest sites and the majority of roost sites were situated in native prairie in moderately to heavily grazed areas. Clayton and Schmutz (1999) and Poulin *et al*. (2005) also observed that crop fields are strongly avoided. In Saskatchewan, the majority of burrow sites examined were found on lacustrine and, to a lesser extent, Solonetzic soils (Harris and Lamont 1985). Nest burrows have been found in the following soil types: loamy sand, silty loam, silty clay loam and sandy loam soils (Dechant *et al.* 2002). Silty loam soils may be more suitable than loamy sand soils for maintaining stable burrows (Dechant *et al.* 2002). Holms *et al.* (2003) found that, in Oregon, burrow reuse was highest in sandier soils; however, livestock trampling resulted in a 24% season-to-season loss of burrows.

In the Great Plains of the United States, burrowing owls have been found to occupy both inactive and active prairie dog colonies (Desmond *et al.* 2000, Dechant *et al.* 2002). However, burrowing owls in larger, well-populated prairie dog colonies typically have higher nest success rates, lower nest predation rates and are more likely to return to their nesting sites (Desmond *et al.* 2000, Dechant *et al.* 2002). Active prairie dog colonies are thought to be preferred as prairie dogs offer an alternate prey source for predators, control the encroachment of dense vegetation and structurally maintain burrows so that they remain suitable for burrowing owls (Dechant *et al.* 2002). Although prairie dogs are not found within the Milk River or South Saskatchewan River watersheds, Richardson's ground squirrels occupy a similar niche and are common in the area. On the Regina Plain in Saskatchewan, burrowing owls selected pastures for nesting that were more level, more likely to be grazed and that had a greater density of Richardson's ground squirrel holes (James *et al.* 1991, Poulin *et al.* 2005). In grassland pastures of the Canadian prairies, Poulin *et al.* (2005) found that burrowing owls preferred sites with higher densities of burrows (11.1 burrows/ha) and selected badger-sized burrows with taller tunnel entrances and soil mounds as opposed to unused burrows.

Burrowing owls typically use several non-nest (satellite) burrows, likely as a predator avoidance strategy (Wellicome 1997, Dechant *et al.* 2002). In central Saskatchewan, an average of six suitable burrows was available within 30 m of the nest burrow (Haug and Oliphant 1990). Habitat fragmentation concentrates burrowing owls into smaller areas of suitable nesting habitat. Increased nesting density can lead to increased intraspecific competition and potentially to higher nest abandonment and lower productivity (Dechant *et al.* 2002).

1.3.3 Foraging Habitat

Burrowing owls tend to remain near the nest burrow during daylight hours, opportunistically foraging, roosting and loafing (Haug and Oliphant 1990). Prey consumed during the day is mainly insects. At night, burrowing owls travel greater distances from their nest site and hunt for small mammals (Haug and Oliphant 1990). Foraging area requirements are therefore considerably larger than nesting area requirements. Nevertheless, foraging areas are still typically located close to nest sites (Stevens and Todd 2008). The nocturnal foraging habits of burrowing owls were investigated using telemetry in an intensively farmed area south of Saskatoon, Saskatchewan (Haug and Oliphant 1990). In this study, peak foraging activity occurred between 2030 and 0639 hours, and 95% of all movements were within 600 m of the nest burrows. Haug and Oliphant (1990) noted that burrowing owls preferred to forage in grass-forb areas in uncultivated fields, ungrazed areas and roadside habitats, and tended to avoid foraging in croplands and heavily grazed pastures. Similar findings were reported by Sissons *et al.* (2001) for a burrowing owl population in southern Saskatchewan which avoided croplands and fallow fields and instead preferred pastures as foraging habitat. Haug and Oliphant (1990) reported that preferred foraging habitats greater than 50 m from nest burrows had dense, permanent vegetation greater than 30 cm but less than 60 cm in height (Haug and Oliphant 1990). Wellicome (1994) noted that these habitats had higher densities of deer mice and meadow voles in comparison to cropland and heavily grazed pastures. Cropland vegetation greater than 1 m in height may be too tall for burrowing owls to effectively forage in due to mobility considerations or prey concealment (Dechant *et al.* 2002). In addition, wheat fields in Canada have been shown to contain a low diversity of small mammals (primarily deer mice), while heavily grazed pastures have been shown to have low relative abundance of prey (Wellicome and Haug 1995). Therefore, while burrowing owls require sparsely vegetated open areas with suitable burrows for nesting and brood-rearing, they also require permanent cover and tall vegetation (30 cm to 60 cm) within their foraging home range to find sufficient prey (Wellicome 1997).

1.3.4 Area Requirements

Nesting area requirements for burrowing owls ranged from 4.1 ha to 7.3 ha in North Dakota (Grant 1965). The minimum foraging home-range size for six radio-tagged burrowing owls in Saskatchewan averaged 241 ha (Haug and Oliphant 1990).

1.4 Burrowing Owl Response to Grazing

The majority of burrowing owl studies suggest grazing is an important factor for maintaining nesting areas. Specifically, the tendency for burrowing owls to nest in moderate to heavily grazed pastures is widely reported across their North American range (Dechant *et al.* 2002). As discussed, Clayton and Schmutz (1999) reported that burrowing owls in southeastern Alberta and Saskatchewan chose moderately to heavily grazed grasslands for nesting and roosting and avoided cultivated fields. In south-central Saskatchewan, Wedgwood (1976) noted that burrowing owls nested in heavily grazed areas within shortgrass pastures containing ground squirrel burrows or American badger excavations. Another study conducted in southern Saskatchewan reported similar results with burrowing owls shown to avoid cropland and fallow fields, preferring instead pastures for nesting (Sissons *et al.* 2001). In Manitoba, the cessation of

grazing is thought to have degraded the suitability of historically successful breeding habitat (Uhmann *et al.* 2001). Moderate grazing is thought to be critical for maintaining consistently short (≤6 cm) vegetation height at nest burrows in this province (Uhmann *et al.* 2001). In North Dakota, burrowing owls nested in moderately and heavily grazed mixedgrass pastures but not in lightly grazed pastures or mowed hayland (Kantrud 1981). Faanes and Lingle (1995) reported that preferred nest sites were in heavily grazed or mowed mixedgrass and shortgrass prairie in the Platte River Valley of Nebraska. Of note, in the arid shortgrass prairies of eastern Colorado, Montana, Wyoming and New Mexico, black-tailed prairie dogs may be able to maintain a low vegetation profile without livestock or bison grazing influences (Holechek *et al.* 2003). Whether the same can be said for Richardson's ground squirrels is a topic that requires further research. Although grazing is recognized as important to the maintenance of suitable burrowing owl nest sites, few studies have been done to investigate which grazing strategies work best to provide sustainable nesting habitat. In addition, few studies have been done to assess the merits of various grazing strategies for maintaining not only nesting but also suitable foraging habitat and healthy populations of prey species.

1.5 Grazing Systems and Burrowing Owl Habitat Management

Table III-5 provides an overview of six pertinent grazing systems and their potential positive and negative effects on burrowing owls and their habitat. A grazing system is a tool used to control the spatial distribution, timing, intensity, and frequency of livestock grazing (Holechek *et al.* 2003). Applied research is needed to properly assess the effects of various grazing systems on burrowing owls in the Milk River and South Saskatchewan River watersheds.

Table III-5 Grazing Systems and Burrowing Owl Habitat Management

Grazing System	Discussion
Continuous (Season-long) Grazing	
Advantages:	One of the primary benefits of a continuous grazing system is that it does not require cross- fencing, and therefore does not create artificial perching sites for burrowing owl raptor predators (Felskie pers. comm.). Clayton and Schmutz (1999) reported that avian predation was responsible for the majority of post-fledgling mortalities in their study area near Hanna, Alberta. Fledgling mortality was high in this study area, estimated at 65% in 1995 and 40% in 1996. Another key benefit of continuous grazing, at light to moderate stocking rates, is that it promotes patchy grazing and therefore creates a more heterogeneous grassland habitat (Robertson *et al.* 1991). Under light to moderate stocking rates, cattle grazing is selective and those areas near water or salt or with more palatable grasses will receive heavier use than other areas. Grazed patches will receive repeated use as these patches have a lesser build-up of litter and higher cover of more palatable re-growth vegetation (Robertson *et al.*1991). As Warnock (1997) suggests, burrowing owl habitat quality can be improved by a mix of short and tall vegetation. Burrowing

	owls require short grass near nests for visibility and taller vegetation within 600 m of nests to provide habitat for their prey (Warnock 1997, Felskie and Scobie pers. comm.). Patchy grazing creates these conditions, with some areas receiving higher use and other receiving light to no use. Habitat patchiness not only provides structural diversity, but also stimulates plant and animal diversity (Rottman and Capinera 1983, Saab *et al.* 1995, Bai *et al*. 2001). This diversity promotes a more varied and likely more stable forage base for burrowing owls. Continuous grazing is better suited to dry mixedgrass and mixedgrass prairie than fescue prairie. Mixedgrass plants are, in general, more tolerant of heavier grazing pressure for sustained periods and earlier in the season than are fescue communities (Adams *et al* 1994). In a continuous grazing system, areas of heavier use can be controlled by the strategic placement of salt, mineral supplements or water. These techniques can be applied to maintain short vegetation near nesting sites, if warranted. However, the appropriate timing and desired intensity of cattle use near burrow sites needs further investigation. Disturbance near burrow sites is likely not desirable during nest initiation and egg-laying (*i.e.*, April to May) as it has the potential to negatively affect nest success (Scobie pers. comm.). Later season use near burrow sites can ensure that short structural conditions are present early the next season, while minimizing trampling risks to nests. Salting is likely not required near nest sites as cattle are naturally attracted to forage near burrow sites. Coppock and Detling (1986) demonstrated that bison fed selectively on moderately grazed, grassy areas near the periphery of prairie dog colonies. These areas were shown to have more readily digestible perennial grasses with higher nitrogen concentrations and accessible green tissue than areas uncolonized by prairie dogs (Coppock and Detling 1986).
Disadvantages:	At high stocking rates, continuous grazing can create uniform, short vegetation across the range. This type of habitat does not offer as many refugia for mammalian burrowing owl prey, such as meadow voles (Wellicome 1997, Scobie pers. comm.). As reported by Haug and Oliphant (1990), heavily grazed pastures were avoided by burrowing owls during nocturnal foraging in their Saskatchewan study area. Intensive grazing not only has the potential to reduce foraging habitat quality, but it also increases trampling risks to nests and chicks. Continuous grazing is also potentially detrimental to riparian systems if alternate water sources are not available. Heavy, persistent use of riparian corridors can diminish their quality as burrowing owl prey habitat (Felskie pers. comm.).

Rotational Grazing - Deferred Rotation	
Advantages:	Deferred grazing in pastures containing active burrowing owl nests may be beneficial to burrowing owls by preventing trampling disturbance at nest burrows during the spring. Deferred grazing can also be advantageous by improving overall plant vigour and range health by preventing defoliation during the critical early season growing stage (Wambolt 1979). Improved range health has numerous benefits to rangeland fauna and flora (Adams *et al.* 2010).
Disadvantages:	If the period of deferral extends into June or July, spring regrowth of vegetation may diminish the quality of burrowing owl nesting grounds by reducing visibility.
Complementary Grazing	
Advantages:	Complementary grazing is a form of season-of-use grazing, with tame pastures grazed earlier or later in the season than native mixedgrass prairie. Complementary grazing, like deferred grazing, can be used to reduce disturbance to burrowing owls during breeding, nest initiation and egg-laying. Tame pasture is well suited for early season use and can maintain its condition if it receives sufficient rest post-grazing. Burrowing owls have been noted to forage and nest in tame pasture with appropriate vegetation structure or small mammal burrows (Dechant *et al.* 2002). Vegetation height should be maintained at between 30 cm to 60 cm to promote optimum foraging conditions (Wellicome 1994). However, if burrowing owls are nesting in tame pasture, maintaining shorter vegetation (less than 10 cm) within 100 m of nesting sites is preferred. No pesticides or herbicides should be applied to tame pastures within 600 m of active burrowing owl nests (Haug and Oliphant 1990).
Disadvantages:	Native prairie habitat generally supports higher species diversity and therefore enriched prey diversity than does monoculture crops (Wellicome and Haug 1995). Conversion of native prairie to tame pasture or cropland contributes to the increasing fragmentation of native prairie habitat, one of the primary limiting factors to burrowing owls in Alberta (Wellicome 1997). For these reasons, complementary grazing systems should ideally be created by converting cropland to tame pasture and not by cultivating native prairie.
Rotational Grazing – Rest Rotation	
Advantages:	Rotational grazing at light to moderate intensity would allow rest and recovery and thereby promote improved grassland range health and suitable foraging habitat for burrowing owls within a 600 m radius of their burrows. A moderate stocking rate would help to promote vegetation patchiness and the retention of variably grazed areas. Enforced periods of rest would improve the health of riparian foraging areas.

Disadvantages:	Rotational grazing could contribute to uniform grazing effects if cattle are allowed to graze the range beyond the 50% utilization point (Kobriger 1980). As discussed, uniform vegetation structure diminishes burrowing owl foraging opportunities. Another disadvantage of rotational grazing systems is the need for cross-fencing, which provides perching sites for predatory raptors (Felskie pers. comm.). If a rotational grazing system is used, deferred rotational grazing or switchback rotational grazing are likely preferable to rest rotation grazing. Rest-rotation systems may not be beneficial for burrowing owls, as the fields that receive a full season of rest will likely not retain the short vegetation that burrowing owls prefer for nest sites (Felskie pers. comm., Kantrud 1981, Clayton 1997, Uhmann *et al.* 2001).
Rotational Grazing – High Intensity Grazing	
Advantages:	Burrowing owls prefer to nest in areas with short vegetation that are frequently grazed (Wellicome 1997). In an intensive grazing system, cattle more evenly utilize a greater percentage of the overall range, increasing the likelihood that burrow sites would be grazed. Encouraging cattle to graze the immediate area around burrows with the use of techniques such as salting or electric fencing would likely be unnecessary under this type of grazing regime. Intensive grazing for short periods could therefore create the structural conditions that are preferred for burrowing owl nest sites; however, this could come at the expense of taller foraging habitat in the area and may cause increased disturbance to nesting burrowing owls. The disadvantages discussed below, therefore, likely outweigh the advantages of intensive grazing.
Disadvantages:	Intensive grazing systems, including short duration or high intensity/low frequency grazing, create increased risks of soil erosion and compaction and can lead to uniform declines in range health throughout the ranch. A general decline in range health would likely not support stable mammal populations that require permanent, taller, denser vegetation for cover, and thereby would diminish burrowing owl foraging habitat. Burrowing owls have been shown to prey preferentially on meadow voles, which have been positively correlated with vegetation height and density (Wellicome 1997). Intensive grazing would also increase trampling risks to burrowing owl chicks and nests.

Riparian Area Grazing	
Advantages:	Well managed riparian habitats are typically productive and harbour rich populations of insects and small mammals (Strand and Merritt 1999). According to a recent study in southern Saskatchewan, burrowing owl nests located near wetlands had a higher nesting success than nests with no wetlands within 2 km (Warnock and Skeel 2002). The proximity and quality of wetland habitat protected from heavy cattle use during sensitive periods may provide an important source of additional prey for burrowing owls (Warnock and Skeel 2002, Felskie pers. comm.). Further research is required to confirm the importance of wetlands and other riparian areas to burrowing owls as a source of forage supply.
Disadvantages:	If the recovery of riparian areas requires the implementation of a rest-rotational grazing system, nesting grounds in fields that receive a full season of rest may require mowing to retain short vegetation near burrowing owl nests.

1.6 Beneficial Management Practice Recommendations

The following general land-use and grazing recommendations provide a variety of means by which to protect or enhance burrowing owl nesting and foraging habitat. Key to burrowing owl conservation is the conservation of native prairie with a heterogeneous mosaic of areas of heavy, moderate and light grazing and areas with suitable burrow escape cover. The recommended BMPs apply to burrowing owl populations within the Milk River and South Saskatchewan River watersheds and throughout the Grassland Natural Region of Alberta (NRC 2006). Further research is required to improve our understanding of the role of grazing in maintaining key habitat and influencing burrowing owl predator and prey species dynamics in the Milk River and South Saskatchewan River watersheds (see Section 1.7). The BMP recommendations discussed below should be reviewed and amended as additional local knowledge becomes available.

Important considerations for managing burrowing owl habitat are:

- Protection traditional burrowing owl nest sites.
- Conservation of native prairie.
- Availability of suitable burrows.
- Maintenance of short vegetation and openness around nest burrows.
- Minimal cross-fencing.
- Promotion of structural heterogeneity across the landscape.
- Maintenance of healthy riparian areas and key nocturnal foraging areas.
- Maintenance of healthy prey populations (AERSD 2012).

1.6.1 General Recommendations

- Protect active and historical (traditional) burrowing owl nesting sites (Haug and Oliphant 1990).
- Conserve large tracts of native prairie where possible to limit further fragmentation of suitable burrowing owl habitat (Warnock 1997).
- Remove marginal cropland from production, where possible, and seed with native species to enlarge habitat patches and reduce fragmentation of burrowing owl habitat (Warnock 1997).
- Control activities within a minimum 600 m radius of the nest burrow, including curtailing the use of pesticides, herbicides and rodent poisoning or eradication methods and minimizing human disturbance (Haug and Oliphant 1990, Dechant *et al.* 2002). Promote organic farming practices.
- Abide by recommended set-back distances and timing restrictions for human activities, including industrial development, near to burrowing owl nests (GoA 2011). Alberta Environment and Parks (AEP) has three separate timing restrictions for burrowing owl nests in the Grassland and Parkland Natural Regions of Alberta. From April 1st to August 15, set-back distances of 200 m, 500 m and 500 m are recommended for low, medium and high disturbance activities. From August 16th to October 15th, nesting sites have set-back distances of 200 m, 200 m and 500 m for the same three levels of disturbances. From October 16th to March 31st, AEP recommends set-back distances of 50 m, 100 m, and 500 m for low, medium and high disturbances.
- In areas with suitable burrowing owl habitat, conduct pre-development surveys to locate burrowing owl nests to plan developments in accordance with AEP set-back guidelines. Ensure wildlife survey data is entered into the Fish and Wildlife Management Information System (FWMIS) database maintained by AEP. Consider reporting ‘zero data’ as well.
- Use Protective Notations under the *Public Lands Act* to protect key burrowing owl habitat and burrows.
- Conserve active Richardson’s ground squirrel colonies in and around active burrowing owl nesting sites. Grounds squirrels maintain suitable burrowing sites for burrowing owls and provide an alternate prey source for potential burrowing owl predators (Desmond *et al.* 2000, Dechant *et al.* 2002).
- Inform landowners about the beneficial ecological roles of American badgers and Richardson’s ground squirrels and about preferred control techniques if control of these species is considered essential.
- Manage active or traditional nesting sites to maintain short (approximately 10 cm) and open, non-woody vegetation conditions within 100 m of nest burrows (Clayton and Schmutz 1999). This can be achieved by regular grazing (see below) or mowing, and by mechanical control of woody species encroachment.
- Encourage interagency cooperation and an ecosystem-based management approach for conserving burrowing owls and their habitat.
- Promote heterogeneous grazing by livestock, ensuring patches that are heavily grazed and patches that are lightly to moderately grazed to allow small mammals to thrive (Marsh 2012).

- Leave narrow strips in cropland as stubble to promote vegetation height and density heterogeneity (Marsh 2012).
- Leave narrow strips in hayfields that are not mown each year to allow cover for small mammals and leave open areas for burrowing owls to forage (Marsh 2012).

1.6.2 Grazing Recommendations

As local site conditions play a large role in determining the best-suited grazing strategy, it is likely that no one grazing system will suffice. Distribution of burrowing owls and their valued habitat components, vegetation type, topography, soil types and existing range health are all factors that must be considered in the selection and implementation of suitable grazing systems. Ultimately, flexible stocking rates and control over cattle distribution are important to the success of any grazing system. Appropriate grazing systems should be developed site-specifically for ranches participating in conservation programs in southern Alberta.

The following is a list of key grazing management recommendations that can be applied to enhance or conserve burrowing owl habitat:

- Encourage regular grazing to maintain short vegetation (less than 10 cm) within 100 m of active or traditional nesting sites. Reduce heavy livestock use at nest sites during the nesting and brood-rearing periods (*i.e.*, May to June) to lessen potential trampling risks.
- Vary stocking rates in accordance with precipitation (*i.e.*, decrease stocking rates in drought years and allow for increased use in wetter years).
- Promote heterogeneous habitat conditions, for example, through patch grazing, with areas of high, moderate and low use and a gradation of short to tall and dense to light residual vegetation and litter cover (Felskie and Scobie pers. comm., Warnock 1997).
- Avoid intensive, high stocking rate grazing systems that encourage uniform utilization of pastures across large areas (Warnock 1997).
- Implement riparian grazing systems to maintain the function, structure and productivity of lotic (flowing water) and lentic (standing water) riparian areas that may serve as foraging sites for burrowing owls (Warnock and Skeel 2002, Felskie pers. comm.).
- Use an appropriate moderate stocking rate and an appropriate percentage of vegetation utilization in order to maintain healthy rangeland within a 600 m radius of nest burrows. This is necessary in order to provide conditions for a stable prey base for burrowing owls. Utilization rates of 25% to 50% are recommended for mixedgrass and dry mixedgrass prairie community types to retain adequate carry-over and maintain range and riparian health (Adams *et al.* 2013a,b).
- Minimize the use of cross-fencing where possible to reduce predatory raptor perch sites near to suitable burrowing owl nesting habitat (Felskie pers. comm.). Experiment with the design of fence post caps to reduce their suitability as raptor perches.

1.7 Research Recommendations

Additional research is needed to compare various grazing systems in mixedgrass and dry mixedgrass prairie and their effects on burrowing owl habitat and population parameters. To adequately assess the effects of grazing management practices over time on rangeland health and

burrowing owl productivity and survival, trends in range health and local burrowing owl population numbers should be monitored. These factors should be assessed in conjunction with monitoring changes in stocking rates, timing of grazing and grazing distribution over time, as well as other relevant land-use and climatic changes. Other factors that should be monitored include causes and rates of burrowing owl mortality, site fidelity and population dynamics, nest site vegetation structure and burrowing owl diet relative to changes in grazing practices. This type of research would assist in the development of preferred grazing strategies to manage for burrowing owl habitat in the Milk River and South Saskatchewan River watersheds. In general, more information is also needed regarding burrowing owl nesting-habitat requirements, diet and nocturnal foraging behaviour in the Milk River and South Saskatchewan River watersheds specifically. Changes in burrowing owl populations and distribution over time in the Milk River and South Saskatchewan River watersheds should also be assessed in relation to changes in Richardson's ground squirrel and American badger populations.

Possible research questions to be addressed include:

- What types of grazing systems, stocking rates or percentages of vegetation utilization are beneficial to burrowing owls?
- How do burrowing owls respond to differences in grazing intensity (light, moderate and heavy grazing) in terms of productivity, nest success and juvenile survival?
- What is the desired timing and intensity of grazing near to nest burrows?
- Does cattle trampling and disturbance near nest burrows contribute to nest failure, juvenile mortality or nest abandonment?
- Does cattle presence at burrowing owl nest sites affect predation rates?
- How does grazing affect small mammal and arthropod prey species abundance and composition in the Milk River and South Saskatchewan River watersheds?
- How important are riparian areas as a source of prey for burrowing owls, and what influence does riparian area grazing have on prey abundance and diversity?
- How do grazing and other land-uses affect burrowing mammal populations (*e.g.*, American badgers and Richardson's ground squirrels) within the Milk River and South Saskatchewan River watersheds? Few studies have researched the habitat preference of ground squirrels or the interaction of cattle grazing and ground squirrel herbivory (Michener pers. comm.).
- What are the potential benefits of prescribed burning as a management tool for maintaining burrowing owl nesting or foraging habitat? How does prescribed burning as a tool compare to grazing?
- How do wet-dry cycles and climate change influence prey availability? (Environment Canada 2012)

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D. LOGGERHEAD SHRIKE

1 INTRODUCTION

1.1 Background

The purpose of this report is to summarize the ecology and habitat requirements of the prairie loggerhead shrike (*Lanius ludovicianus excubitorides*) in southern Alberta. Based on this information and supporting scientific studies, various grazing systems are compared in terms of their potential implications to loggerhead shrike ecology and habitat. This discussion is followed by a summary of recommended beneficial management practices to enhance loggerhead shrike habitat in the Milk River and South Saskatchewan River watersheds in Alberta, with broader application to the range of this species within the Grassland Natural Region of Alberta (Natural Regions Committee [NRC] 2006). Lastly, a brief summary of additional information needs that would improve our understanding of loggerhead shrike ecology, habitat needs and response to land management is presented.

The loggerhead shrike is a predatory songbird with a grey body and a characteristic black facial mask (Semenchuk 1992). As both a passerine and a top-level predator, the loggerhead shrike occupies a unique position in the food chain. The hooked bill and tomial 'tooth' of the loggerhead shrike are functionally similar to the notched bills of falcons, enabling them to be predators of vertebrates and setting them apart from other songbirds. The colloquial name 'butcher bird' is often used to refer to the loggerhead shrike and is in reference to the species' habit of impaling larger prey items on sharp objects, such as thorns and barbed wire (Committee on the Status of Endangered Wildlife in Canada [COSEWIC] 2014).

The breeding range of the prairie subspecies of the loggerhead shrike extends from southeastern Alberta to southwestern Manitoba and south through the Great Plains to Mexico (COSEWIC 2014). The core of the species breeding range in Alberta includes the northern half of the Grassland Natural Region and the southern half of the Parkland Natural Region (*i.e.*, from Brooks, Hanna and Stettler eastward to the Saskatchewan border) (Prescott 2009, 2013).

Loggerhead shrikes are ranked as 'Sensitive' in Alberta under the General Status listing (Government of Alberta [GoA] 2016). The species is listed as a species of Special Concern under the provincial *Wildlife Act*. Nationally, the loggerhead shrike was assessed by COSEWIC in 1986 and given the status of 'Threatened' (Cadman 1986). That status was re-confirmed in 2004 and again in 2014, and the species remains 'Threatened' in 2016 (COSEWIC 2004, 2014, 2016). In 2005, the loggerhead shrike was added to Schedule 1of the federal *Species at Risk Act*. Loggerhead shrikes and their eggs and nests are also protected from harm under the federal *Migratory Birds Convention Act*. Internationally, the loggerhead shrike is designated as a species of 'Least Concern' by the International Union for Conservation of Nature and Natural Resources (IUCN) (BirdLife International 2012).

The range of the loggerhead shrike in Alberta has declined over the past few decades (Prescott and Bjorge 1999, Prescott 2013), and shrike populations have suffered up to 80% declines due to fragmentation of breeding grounds, habitat loss due to conversion of native prairie to cropland,

accumulation of toxins from pesticide residues consumed through their prey and possibly loss and degradation of wintering habitat (Johns *et al.* 1994, Prescott and Bjorge 1999, COSEWIC 2004, 2014). Telfer (1992) found that areas of Alberta and Saskatchewan with the highest declines in loggerhead shrike populations in recent decades had lost 30% of their unimproved pasture (native prairie) between 1946 and 1986. Areas with a fairly stable loggerhead shrike population had lost a significantly lesser amount (12%) of native prairie to cropland production (Telfer 1992).

Loggerhead shrike population surveys have been completed in Alberta approximately every five years since 1987. The most recent surveys were completed in 2003, 2008 and 2013 (Prescott 2004, 2009, 2013). In 2003, observers recorded 168 loggerhead shrikes at 144 unique road-side locations, resulting in an estimated linear density of 1.78 pairs/100 km of road (Prescott 2004). In 2008, observers recorded 151 loggerhead shrikes at 121 unique locations, for a linear density of 1.54 pairs/100 km of road (Prescott 2009). In 2013, a total of 158 loggerhead shrikes were observed at 130 locations, resulting in a linear density of 1.61 pairs/100 km of road (Prescott 2013). Highest loggerhead shrike densities in all three surveys were observed in the east-central portion of the province. These three loggerhead shrike surveys resulted in provincial population estimates of 8,327, 7,721 and 7,508 pairs, respectively. The current Alberta loggerhead shrike population is estimated at 3,000 pairs (GoA 2016).

A national Recovery Strategy was prepared for the loggerhead shrike in 1994 (Johns *et al.* 1994). Several studies have subsequently been conducted to examine the ecology of loggerhead shrikes in Alberta (Prescott and Collister 1993, Collister 1994, Collister and Henry 1995, Bjorge and Prescott 1996). The national Recovery Strategy was updated in 2015 (Environment Canada 2015). It summarized the current status of the loggerhead shrike based on the most recent COSEWIC assessment, documented threats, mapped critical habitat and outlined the ongoing recovery efforts.

1.2 Ecology

Loggerhead shrikes return to their Alberta breeding grounds between late March and early April from wintering grounds in the southern United States and Mexico. Based on research in Alberta, loggerhead shrikes typically initiate egg-laying in mid-May and lay an average of six eggs, which are incubated from 14 to 20 days (Smith and Bjorge 1992, Collister 1994). Male loggerhead shrikes are responsible for providing female shrikes with food during the incubation period as only female shrikes incubate the eggs. In southeastern Alberta, peak hatching occurs between June 2 and 10 (Collister 1994). The nestling period lasts approximately 17 to 20 days. Collister (1994) reported nest success rates of 48.7% (n=36) for 1992 and 37.8% (n=28) for 1993 for a loggerhead shrike population in southeastern Alberta. Yosef (1996) reported an overall nest success of 56% for 2034 nests from various studies throughout the range of loggerhead shrikes. Loggerhead shrikes have been noted to renest following failure of the first nesting attempt (Collister 1994, Yosef 1996). Collister (1994) found that replacement nests were typically built within 100 m of the first nest. Studies in Alberta found that 32% of adult loggerhead shrikes returned to the same site in subsequent years, while only 1.2% of juveniles returned to the nest site (Collister and De Smet 1997). Downey and Taylor (2003) found that six out of 11

loggerhead shrikes in the Milk River Watershed were located in or adjacent to previously known shrike territories.

1.2.1 Diet

Loggerhead shrikes are opportunistic predators that prey on small mammals, birds, reptiles, amphibians, occasionally carrion, and insects such as grasshoppers, beetles and bees (Dechant *et al.* 2002). Loggerhead shrikes are passerines and so do not have raptorial feet to handle live prey. As a result, loggerhead shrikes have evolved the unique ability to impale prey on sharp objects such as thorns and barbed wire (Yosef 1996). Invertebrates constitute the majority (approximately 70%) of the loggerhead shrike's diet during the breeding season; however, shrikes have been known to adjust their diet according to local prey availability (Prescott and Bjorge 1999). In Alberta, vertebrate prey commonly consumed by loggerhead shrikes include thirteen-lined ground squirrels (*Spermophilus tridecemlineatus*), meadow voles (*Microtus pennsylvanicus*) and sagebrush voles (*Lagurus curtatus*) (Prescott and Bjorge 1999).

1.2.2 Predators

Predators of loggerhead shrike eggs and young include black-billed magpies (*Pica hudsonia*), bull snakes (*Pituophis melanoleucus*), long-tailed weasels (*Mustela frenata*), foxes (*Vulpes* spp.), common raccoons (*Procyon lotor*) and feral cats (*Felis domesticus*) (Collister 1994, Gawlik and Bildstein 1990, Collister and Wilson 2007, COSEWIC 2014).

1.3 Habitat Requirements

1.3.1 General

Loggerhead shrikes prefer flat, open habitats with scattered clumps of shrubs or hedgerows and are often found close to a variety of habitat types such as pastures, meadows, farmsteads and railroad rights-of-way (Brooks and Temple 1990, Bjorge and Prescott 1996, Dechant *et al*. 2002). In Alberta, thorny buffaloberry (*Shepherdia argentea*), willow (*Salix* spp.), or common caragana (*Caragana arborescens*) typically form the woody component of loggerhead shrike habitat. A survey of loggerhead shrike populations in the Milk River Watershed found that out of eleven sites occupied by shrikes, native grassland was the most abundant habitat type within 400 m of loggerhead shrike sightings (shrike sites). Native prairie made up an average of 43% of the surrounding habitat within shrike sites, while dry land cultivation comprised 34.1% and tame pasture made up 7.3% of surrounding habitat (Downey and Taylor 2003). Six out of the 11 sites corresponded with Collister's (1994) findings that hedgerows/shelterbelts were used more frequently than single shrubs in the Milk River area. No sites occupied by loggerhead shrikes were found to contain greater than 30% shrub cover (Downey and Taylor 2003).

As part of the Multiple Species at Risk (MULTISAR) conservation program in southern Alberta, a Habitat Suitability Index (HSI) model was developed for loggerhead shrike habitat use in the Milk River Watershed (Downey 2004a). The final HSI included four variables: shrub cover; graminoid cover; slope; and farmyards. According to the HSI model, quarter-sections containing at least 80% graminoids and 5% shrubs on flat terrain represent ideal loggerhead shrike habitat. Slopes greater than 15° and shrub cover greater than 30% decreases the potential suitability of an

area for loggerhead shrikes. Farmyards increase the HSI value due to the higher potential for these sites to have shrubs (hedgerows) and edge habitat.

A second HSI model was developed by MULTISAR for loggerhead shrike habitat use in the headwaters of the Oldman River Watershed (Downey 2004b). Again, four variables were included in the final model, including shrub cover; graminoid cover; tree cover; and slope. According to this model, quarter-sections that contain at least 80% graminoids, 20% or less trees, and 5% to 30% shrubs on flat terrain would be ideal loggerhead shrike habitat. Conversely, as slope, tree cover and shrub cover increase, suitability of the habitat for loggerhead shrikes decreases. Variables considered for the final model but not included due to a lack of data for the areas included farm yards, grass height and shrub complexes.

Also as part of the MULTISAR program, a Resource Selection Function (RSF) model was developed for loggerhead shrike habitat use in southern Alberta (Skilnick *et al.* 2010). RSFs are thought to be more robust than HSI models due to their rigorous and empirical evaluation of model performance using statistics (Boyce *et al.* 2002). RSFs estimate relative probability of occurrence by species of concern on the landscape based on various physical and anthropogenic variables. The best-fit RSF model as determined by Skilnick *et al.* (2010) contained 13 variables: elevation; slope; distance to linear vegetation; distance to point vegetation; distance to cultivation; distance to tame pasture; distance to preferred shrub cover; distance to rural areas (farm yards); distance to lentic wetlands; percent shrub and non-vegetation cover; amount of edge habitat; and distance to railways. The variable distance to point vegetation had the strongest relationship with presence of loggerhead shrikes on the landscape. Loggerhead shrikes were also usually found closer to preferred amounts of shrub cover as well as closer to shrubs of a preferred height (>1 m). Unexpectedly, loggerhead shrikes were found further from linear vegetation features such as hedge rows. Loggerhead shrikes also appear to be selecting for flat terrain; lower elevations; areas with less edge habitat; further distances from tame pastures/hay fields and railways; and closeness to lentic wetlands, cultivation, farm yards and railways.

1.3.2 Nesting Habitat

Loggerhead shrikes prefer to nest in shrubs 2 m to 3 m above the ground surface (Environment Canada 2015). Wershler (1989) conducted a study of loggerhead shrikes in the Milk River region and found that for nesting, 20% used thorny buffaloberry, 33% willow, 20% common caragana, 13% Manitoba maple (*Acer negundo*), 7% Siberian elm (*Ulmus pumila*) and 7% silver sagebrush (*Artemisia cana*). Of the nest sites found in the Milk River area, 57% were found primarily in natural shrub communities in valleys, including the Milk River, and in scattered upland sites, and 43% of nest sites were found in exotic shelterbelts and hedgerows in cultivated areas (Smith 1991). Hellman (1994) noted that loggerhead shrikes tended to avoid nesting in common caragana in her Manitoba study area and preferred to nest in trees with wider canopies and larger diameters than non-nest trees. Loggerhead shrikes were found to prefer shrubs taller than 1.8 m when choosing a shrub or tree for nesting (Shen *et al.* 2013). Hellman (1994) also noted that nest sites with lower amounts of understory had highest nest success.

1.3.3 Foraging Habitat

Loggerhead shrikes use dead trees, tall shrubs, utility wires and fences as perches for hunting (Yosef 1996, Prescott and Bjorge 1999). Yosef and Grubb (1994) identified availability of hunting perches as a limiting factor for loggerhead shrike habitat. Loggerhead shrikes will impale their prey on the thorns of thorny buffaloberry or other thorny shrubs, sharp twigs or barbed wire (Yosef 1996, Prescott and Bjorge 1999). Collister (1994) found that loggerhead shrikes in southeastern Alberta preferred to forage in native prairie and tame pasture and avoided cereal crops and right-of-way habitats. Although the number of forages observed in railroad right-of-way habitats was lower than expected, Collister (1994) noted that loggerhead shrikes still made 722 (28%) foraging attempts and experienced the highest foraging success in this type of habitat. Collister (1994) speculated that the taller (greater than 30 cm), dense vegetation of rights-of-way may be important reserve areas for vertebrate prey when arthropod prey is scarce. Prescott and Collister (1993) found that sites occupied by loggerhead shrikes in southeastern Alberta had a greater amount of tall grass (>20 cm) than did sites that were unoccupied.

In other parts of its range, researchers have highlighted the importance of short grass and grazing activity in habitat selection by breeding loggerhead shrikes (Prescott and Bjorge 1999). Grassy areas provide habitat for insects, a key forage species during the breeding season, and shorter grass would allow easier detection and capture by loggerhead shrikes. In Alberta, as Prescott and Bjorge (1999) note, the preference of loggerhead shrikes for “taller” grass may not be inconsistent with the apparent preference for “shorter” grass in eastern and central North America, as Alberta has a more arid climate and naturally much shorter grass heights than in more easterly parts of the continent.

Yosef and Grubb (1993) examined the effect of mowing within loggerhead shrike territories on the hunting behaviour and diet of loggerhead shrikes. Loggerhead shrikes were seen to adjust their hunting behaviour in mowed and unmowed areas; however, the rate of prey capture and the range of species captured did not vary between mowed and unmowed territories. Loggerhead shrikes in unmowed areas had increased flight time and shifted from ground hunting to aerial chase or hunting from a hover, resulting in an overall increase in net energy expenditure.

1.3.4 Area Requirements

Loggerhead shrike breeding territories along a railroad-right of way in southeastern Alberta were found to be asymmetric and averaged 8.5 ha (Collister 1994). Mean territory size of loggerhead shrikes is 5.9 ha during the incubation period and 6.7 ha during the nestling period (Collister 1994). Collister (1994) noted that the mean maximum radius of excursion from loggerhead shrike nests over two years ranged up to 360 m.

1.4 Loggerhead Shrike Response to Grazing

Cattle grazing has the potential to impact loggerhead shrikes directly and indirectly. Rubbing or browsing by cattle on shrubs used for nesting can cause nest destruction or abandonment (Pittaway 1991). Cattle herbivory affects the availability and composition of loggerhead shrike prey species by altering vegetation structure and plant species composition.

Few studies have compared the merits of various grazing systems for loggerhead shrikes. However, several studies in the United States have assessed the response of loggerhead shrikes to various intensities of grazing. In the shrubsteppe habitats of Nevada, Idaho and Utah, studies have shown that loggerhead shrikes have either increased in abundance or have not been affected by heavy to moderate grazing (Saab *et al.* 1995). Similarly, in Virginia and South Carolina, Luukkonen (1987), Gawlik (1988) and Gawlik and Bildstein (1990) suggest that grazed areas are preferred loggerhead shrike habitat. Blumton (1989) found that loggerhead shrike productivity was highest in areas with grasses of medium height (9.1 cm to 18 cm) in comparison to similar areas with shorter or taller grass. However, as noted by Prescott and Collister (1993), heavy grazing may be limiting to loggerhead shrikes in southern Alberta since loggerhead shrikes were found to occupy habitats with a higher percentage of tall (≥ 20 cm) grass and avoided habitats with shorter grass (<20 cm). This observation may have been an indication of greater prey abundance in taller grass habitats; however, further research is needed to investigate the relative abundance of prey and the frequency of foraging by loggerhead shrikes in grazed versus ungrazed grasslands. Mills (1979) reported that non-breeding loggerhead shrikes preferred to forage in areas of short grass, but would feed in taller vegetation if prey availability was higher. In Arizona, Bock *et al.* (1984) observed no significant difference in loggerhead shrike numbers in winter or summer in native prairie that had been continuously grazed at moderate intensity compared to grassland that had been rested for more than 10 years.

1.5 Grazing Systems and Loggerhead Shrike Habitat Management

Table III-6 provides an overview of five pertinent grazing systems and their potential positive and negative implications to loggerhead shrikes and their habitat. A grazing system is a tool used to control the spatial distribution, timing, intensity, and frequency of livestock grazing (Holechek *et al.* 2003). Applied research is needed to properly assess the effects of various grazing systems on loggerhead shrikes in the Milk River and South Saskatchewan River watersheds.

Table III-6 Grazing Systems and Loggerhead Shrike Habitat Management

Grazing System	Discussion
Continuous (Season-Long) Grazing	
Advantages:	Continuous grazing at moderate stocking rates can be beneficial to loggerhead shrikes if it promotes patchy grazing, with some areas lightly grazed and other areas more heavily grazed. This type of grazing pattern creates heterogeneous habitat that promotes refugia for small mammals and also stimulates plant species diversity and, in turn, insect diversity. As loggerhead shrikes prey primarily on insects during the summer, grazing systems that stimulate insect diversity but do not promote damaging “pest” species are desirable. Based on a study of grasshopper species in the mixedgrass and fescue prairie in southern Alberta, Hardman and Smoliak (1982) concluded that the majority of grasshopper species that can cause the most damage to rangeland preferred sparsely vegetated habitats such as heavily grazed range. Jonas *et al.* (2002), in their study of the response of

	arthropods to various land- use types in central Kansas, concluded that the diversity and richness of all arthropod groups examined (specifically Coleoptera [beetles] and Orthoptera [crickets and grasshoppers]) were positively correlated with plant species diversity and richness. Willoughby (1993) noted that plant species diversity was higher in moderately grazed areas than in overgrazed or ungrazed areas in fescue prairie in southwestern Alberta.
Disadvantages:	Intensively stocked continuously grazed systems can have a detrimental impact on the quality and availability of suitable shrubs for nesting. Under continuous grazing systems, cattle pressure on wetlands and small riparian zones is typically increased. Persistent grazing during the dormant season will progressively set back woody species (Fitch and Adams 1998). Heavy browsing can deplete root reserves and inhibit the establishment and regeneration of woody species, ultimately leading to invasion by disturbance or weedy species (Fitch and Adams 1998). Physical damage by rubbing and trampling, if persistent, can also kill off woody vegetation. The die-off or decline in cover values of shrubs and willows reduces the number and quality of suitable nesting sites for loggerhead shrikes.
Complementary Grazing	
Advantages:	Complementary grazing, a form of rotational grazing, may be beneficial to loggerhead shrikes by allowing for deferred grazing of native prairie until the end of June, with tame pasture grazed earlier in the season. This delay allows loggerhead shrikes to complete nesting and fledge young in shrubs near wetlands or riparian areas or in upland native prairie prior to cattle grazing in these areas. Late-season grazing (*i.e.*, after July) of mixedgrass prairie is preferred over early-season grazing to allow for improved ground cover and litter accumulation, particularly for areas that have been heavily grazed (Naeth *et al.* 1991).
Disadvantages:	In his study of loggerhead shrikes in southern Alberta, Collister (1994) noted that the majority of shrikes nested within native prairie. Habitat within territories averaged approximately 52% native prairie and only 8% tame pasture. Loss of native prairie habitat to tame pasture to facilitate complementary grazing would therefore not be beneficial to loggerhead shrikes.

Rotational Grazing – Rest Rotation and Deferred Rotation	
Advantages:	One of the immediate benefits of rotational grazing systems to loggerhead shrikes is an increase in the availability of suitable perch sites. Yosef and Grubb (1994) found that habitat suitability for loggerhead shrikes in their Florida study area was enhanced by introducing hunting perches such as fence posts. The addition of fence post perches allowed loggerhead shrikes to forage in previously unsuitable areas and thereby reduce the size of their territories. Reducing the size of existing territories may allow more territories to be fitted into a given area and thereby increase the size of a local population (Yosef and Grubb 1994). A reduction in territory size may have associated benefits to the nutritional condition of adult loggerhead shrikes. Smaller territories mean loggerhead shrikes expend less energy in defense of a larger, unused area and capitalize on nutritional gains from a smaller, better utilized area that is more easily defended (Yosef and Grubb 1994). Yosef and Grub (1994) noted that significantly more young were fledged in manipulated territories. Deferred rotational grazing systems may also improve the prey base available to loggerhead shrikes by improving the overall condition of the range by allowing for periodic rest and recovery and improved litter cover. Periodic seasonal rest also allows for improved health of shrub vegetation by ensuring that it is not consistently grazed at the same period each year.
Rotational Grazing – High Intensity Grazing	
Advantages:	Heavy grazing can lead to an increase in the density of grasshoppers, a preferred prey item of loggerhead shrikes. In their study of fescue grasslands at the Stavely Research Station in southern Alberta, Holmes *et al.* (1979) collected more grasshoppers from heavily and very heavily grazed fields than from light and moderately grazed fields.
Disadvantages:	Intensive grazing systems force cattle to be less selective when grazing and encourages more even distribution of grazing across the range. Cattle will graze all types of available forage, including shrubs and woody browse, which would not be ordinarily favoured. This creates obvious pressures on shrubs used for nesting by loggerhead shrikes. Intensive grazing systems are also typically detrimental to the viability of riparian areas and associated loggerhead shrike nesting habitat. In addition, intensive grazing for prolonged periods may be detrimental to the availability of small mammal prey (Fagerstone and Ramey 1996).

Riparian Area Grazing	
Advantages:	Although loggerhead shrikes are primarily found on flat prairie, small riverine riparian zones are likely beneficial in providing them with nesting sites if they contain large flat valleys with shrubs, such as the Lost River area in southeastern Alberta (Downey 2004a). Wetland riparian areas are especially important for loggerhead shrikes in the drier areas of southeastern Alberta due to their higher potential for supporting shrub growth (Downey 2004a). Riparian area grazing systems, including rotational grazing, corridor fencing and riparian pastures, are all geared toward protecting the health and integrity of riparian systems. Protecting the vitality of deeply-rooted woody species such as willows and large shrubs such as thorny buffaloberry is vital to maintaining bank stability, trapping sediment and reducing erosion. Therefore the goals of riparian area management are consistent with wildlife habitat goals such as providing nesting and foraging habitat for loggerhead shrikes.
Disadvantages:	In the absence of well-managed riparian area grazing, cattle will tend to linger in these areas and cause degradation of riparian area vegetation and overall riparian health. As discussed, heavy utilization of shrubs and trees in riparian areas prevents the establishment of seedlings and can ultimately eliminate woody vegetation entirely from these sites. This has obvious negative implications to loggerhead shrike nesting and foraging opportunities. Planning suitable riparian area grazing systems, however, should not come at the cost of degrading upland prairie range health. The design of riparian area grazing strategies must therefore consider management of the overall rangeland unit.

1.6 Beneficial Management Practice Recommendations

The following general land-use and grazing recommendations provide a variety of means by which to protect or promote loggerhead shrike nesting and foraging habitat. The suggested beneficial management practices apply to loggerhead shrike populations within the Milk River and South Saskatchewan River watersheds and throughout the Grassland Natural Region of Alberta. The recommendations are based on the current knowledge of loggerhead shrike ecology and habitat use in the Milk River and South Saskatchewan River watersheds as well as its greater breeding range in North America. Further research is required to improve our understanding of the influence of grazing as well as loggerhead shrike habitat requirements in the Milk River and South Saskatchewan River watersheds (see Section 1.7). The beneficial management practice recommendations should be reviewed and amended as additional knowledge becomes available.

1.6.1 General Recommendations

Native Prairie Conservation

- Conserve remaining native prairie (Telfer 1992). Where land has been cultivated, maintaining tame pasture is preferred to cropland (Telfer 1992, Dechant *et al*. 2002).
- Reclaim or restore cultivated areas to native grassland wherever possible.
- Within native prairie, protect silver sagebrush (Woods and Cade 1996) and upland shrubland habitats (Downey 2004a, b).
- Discourage the use of herbicides to control shrub re-growth in native prairie fields and tame pastures.
- Maintain graminoid and herbaceous cover dominance versus dense woody species encroachment (Brooks and Temple 1990, Downey 2004a). Five percent shrub cover on flat terrain is considered preferred loggerhead shrike habitat in the Milk River Watershed (Downey 2004a). Where woody cover exceeds 30% in formerly suitable loggerhead shrike habitat, investigate the use of prescribed burning or trimming and manual removal of shrubs and trees as opposed to the use of herbicides or frequent mowing (Yosef 1996).

Riparian Area Conservation

- Protect lotic (flowing water) and lentic (standing water) riparian woody edge habitat by managing for high riparian health.

Habitat Enhancements

- Protect abandoned railroad rights-of-way and undertake habitat enhancements in these areas (*e.g.*, through native shrub plantings) where necessary (Collister 1994). Thorny buffaloberry is a preferred shrub for loggerhead shrikes.
- Plant willow, thorny buffaloberry or other like native shrubs in areas where these species have been removed or in areas where woody species are not naturally regenerating due to heavy browsing pressure (Telfer 1992, Dechant *et al.* 2002). Planting of shrubs is recommended particularly in areas with high vegetation diversity (*e.g.*, along the border of tame pasture and native prairie and road allowances) and near fences (Telfer 1992, Bjorge and Prescott 1996). Telfer (1992) recommends planting one patch of thorny buffaloberry or willow per quarter-section in suitable locations as a means to improve loggerhead shrike habitat.
- Maintain shelterbelts around old farmsteads and field edges (Collister 1994, Dechant *et al.* 2002). Investigate the benefits of diversifying shelterbelts by planting native thorny shrubs such as thorny buffaloberry or hawthorn (*Crataegus* spp.) as well as planting or leaving 2 m to 4 m wide grassy areas along shelterbelts to increase foraging areas near nests (Hellman 1994).
- To reduce the potential for increased predation pressure due to systematic sampling, use means to reduce the linear nature of shelterbelts and avoid planting shrubs in linear strips along fencelines (Yosef 1994). For example, plant multiple, irregular patches of shrubs or trees along shelterbelts and create larger blocks of habitat adjacent to strips of woody vegetation (Dechant *et al.* 2002).
- Preserve tree and shrub growth in abandoned farm yards (Bjorge and Prescott 1996).

Pest Control

- Reduce or avoid the use of insecticides and organochlorides to avoid contamination of loggerhead shrike prey species (Dechant *et al*. 2002).
- Limit the use of pesticides in known loggerhead shrike habitat. Although the exact effects of pesticides on loggerhead shrikes are not known, as a top-level predator pesticide will bioaccumulate in loggerhead shrikes. Pesticides are believed by some to be at least partly responsible for the decline of loggerhead shrikes (Fraser and Luukkonen 1986).

Population Monitoring

- Continue to monitor loggerhead shrike populations in Alberta every five years. Population trend, distribution, productivity and habitat use should be monitored (Environment Canada 2015).
- Submit loggerhead shrike observations to Alberta Environment and Parks for inclusion in the Fish and Wildlife Management Information System (FWMIS) database. Consider submitting 'zero data' for locations that were searched for loggerhead shrikes but in which none were found.

1.6.2 Grazing Recommendations

Appropriate grazing systems for loggerhead shrikes should be developed site-specifically for participating ranches as part of the MULTISAR conservation program in the Milk River and South Saskatchewan River watersheds.

Grazing systems that will benefit loggerhead shrikes should perform the following functions:

- Promote plant and insect species diversity.
- Provide habitat for small mammals.
- Provide areas of undisturbed or minimally disturbed shrubby nesting habitat in relatively flat sites (upland or lowland) during the breeding season.

To provide these characteristics, the following grazing management principles should be applied:

- Promote heterogeneous vegetation heights by encouraging light to moderate grazing at suitable safe-use factors for native grassland plant communities (*i.e.*, 25% to 50%) (Adams *et al.* 2005, 2013a, b).
- Retain areas of taller grass (>20 cm) near to suitable loggerhead shrike nesting habitat to serve as food reserves for small mammals (Prescott and Collister 1993).
- Manage cattle grazing in riparian areas and near to shrubs used for nesting by using fencing or timing use of these areas to avoid the peak nesting season (*i.e.*, mid-May to mid-June). Note that as female loggerhead shrikes sometimes mate with more than one male or switch mates, managers should aim to provide suitable breeding areas large enough to support several, asymmetrically-shaped, average-sized territories (approximately 5.9 ha to 6.7 ha/territory) (Haas and Sloane 1989, Collister 1994).
- Conduct riparian health assessments to determine the composition, age class, canopy cover and percent utilization of woody species. Implement riparian grazing systems where necessary to improve riparian health and thereby improve cover of woody species.

1.7 Research Recommendations

Further research is required to compare loggerhead shrike productivity and survival under various grazing systems and to evaluate the specific benefits and possible detriments of grazing on loggerhead shrikes. For example, studies could be conducted that:

- Examine differences in loggerhead shrike invertebrate and vertebrate prey abundance and composition in light, moderate and heavily grazed grasslands in the Milk River and South Saskatchewan River watersheds.
- Evaluate the frequency of foraging by loggerhead shrikes in grazed versus ungrazed grassland.
- Compare loggerhead shrike productivity and other population parameters in habitat managed under continuous, rotational and complementary grazing systems.
- Conduct an inventory of shelterbelts in the province to identify potential habitat for the loggerhead shrike. The provincial Grassland Vegetation Inventory, an ecological site mapping database for the White Zone, includes linear features such as shelterbelts. This database is freely available from the provincial government and can be used to assist in identifying potential loggerhead shrike habitat.
- Conduct field verification of the MULTISAR RSF model.
- Use emerging technologies to identify loggerhead shrike wintering areas (Environment Canada 2015).
- Assess potential threats of habitat loss and intra-specific competition on wintering grounds (Environment Canada 2015).
- Assess the effects of pesticides on loggerhead shrike mortality and prey availability (Environment Canada 2015).
- Determine the effect of diseases such as West Nile virus on loggerhead shrike population trends (Environment Canada 2015).
- Determine the effect of road mortality on loggerhead shrike populations (Environment Canada 2015). Mortality from vehicle collisions may be a significant factor affecting loggerhead shrike populations due to the species' preference for habitat features often associated with roads (*e.g.*, fencelines, utility lines) (Luukkonen 1987, Flickinger 1995).
- Assess the importance of predation as a threat to loggerhead shrike survivability (Environment Canada 2015).
- Improve the assessment of the effects of local and continental weather on shrike survivability and productivity (Environment Canada 2015).

Additional studies are needed to evaluate differential nesting success of loggerhead shrikes in natural shrub communities versus shrikes nesting in exotic linear shelterbelts and farmyard hedgerows in the Milk River and South Saskatchewan River watersheds. This type of study would evaluate whether linear shelterbelts are searched systematically by predators or are subject to higher predation pressure due to concentration of prey species, greater edge habitat and greater diversity of predatory species (Yosef 1994, Haas 1997). The goal of this research would be to determine whether shelterbelts constitute "sink" versus "source" habitats for loggerhead shrikes (Yosef 1994). This information is important to evaluate the quality of shelterbelts as s loggerhead shrike nesting habitat in comparison to natural shrubby corridors.

To more thoroughly determine the preferred types and structural characteristics of suitable nest shrubs and associated vegetation, more detailed information on loggerhead shrike habitat within the Milk River and South Saskatchewan River watersheds needs to be gathered. This includes acquiring information on grass height, litter cover, and shrub species type, height, canopy cover, percent utilization and basal diameter. Nesting habitat information should be analyzed in relation to loggerhead shrike productivity data.

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E. GRASSLAND BIRD GROUP 1 (MOUNTAIN PLOVER AND MCCOWN'S LONGSPUR)

1 INTRODUCTION

The purpose of this report is to summarize the ecology and habitat requirements of the mountain plover (*Charadrius montanus*) and McCown's longspur (*Calcarius mccownii*) in southern Alberta. These two species are grouped together in this report based on their preference for similar habitat types (*i.e.*, sparsely vegetated, dry mixedgrass prairie). Based on information on the species' ecology and habitat preferences, livestock grazing interactions and the potential effects of various grazing systems on mountain plovers, McCown's longspurs and their habitats are discussed. This discussion is followed by a summary of recommended beneficial management practices (BMPs) to enhance habitat for these species in the Milk River and South Saskatchewan River watersheds in Alberta and throughout their range in the Grassland Natural Region of Alberta (Natural Regions Committee [NRC] 2006). Lastly, a brief summary of additional research needs is presented.

2 MOUNTAIN PLOVER

2.1 Background

The mountain plover is a slender-billed, long-legged upland plover with pale body and inconspicuous plumage (Semenchuk 1992). It is similar in size to a killdeer (*Charadrius vociferus*), but it lacks the black breast bands and rust-coloured rump of that species (Semenchuk 1992). The mountain plover, contrary to its name, avoids shorelines and montane areas and is primarily associated with short-grass prairie that has been burned or heavily grazed (Alberta Sustainable Resource Development [SRD] 2003). The species is considered a primary endemic species to the Great Plains, occurring exclusively on the prairies of North America (Knopf 1996a). Bison (*Bison bison*), pronghorn (*Antilocapra americana*) and prairie dog (*Cynomys ludovicianus*) grazing in combination with fire were historically important for maintaining suitable habitat for this species (Knopf 1996a). The mountain plover is a regular transient visitor to Alberta, but is at the extreme northern periphery of its breeding range in southeastern Alberta (Knopf and Wunder 2006). The species is found only in the Grassland Natural Region in the southeastern part of the province (Semenchuk 1992, Natural Regions Committee [NRC] 2006, Federation of Alberta Naturalists 2007).

Mountain plovers are listed as 'At Risk' in Alberta due to their small and sporadic population size (zero to six pairs), limited distribution and narrow habitat preferences (Government of Alberta [GoA] 2016). The species is listed as a 'non-game animal' under the provincial *Wildlife Act*, making it illegal to kill, possess, buy or sell them. The mountain plover will be up-listed to 'Endangered' under the *Wildlife Act* in 2016 (GoA 2016). Due to the rarity of mountain plovers in Canada, they were designated as 'Endangered' nationally by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) in 1987 (COSWEIC 2009). The species remains 'Endangered' nationally (COSEWIC 2016) and were listed under Schedule 1 of the federal *Species at Risk Act* (*SARA*) in 2003. Mountain plovers and their eggs and nests are also protected

from harm under the national *Migratory Birds Convention Act*. Internationally, the mountain plover is listed as 'Near Threatened' by the International Union for Conservation of Nature and Natural Resources (IUCN) (BirdLife International 2012).

The population of adult mountain plovers in Canada has been estimated at less than 50 for the past 25 years (Wershler 2000). The maximum number of individuals recorded in Canada in one year is 11 individuals (COSEWIC 2009). Few historical accounts are available that provide an accurate indication of what the pre-settlement mountain plover population may have been in Canada (SRD 2003). In 2000, the current North American population of mountain plovers was estimated to range from 8,000 to 10,000 birds (Wershler 2000). In 2006, Knopf and Wunder (2006) revised the estimated global population to between 10,000 and 19,000, with a trend showing a decreasing population.

Present range management practices that discourage heavily grazed grassland are thought to restrict suitable breeding habitat for mountain plovers in Canada (Wershler 2000). The species prefers low vegetation generally less than 10 cm in height (Knopf and Wunder 2006). There are only two known breeding sites of mountain plovers in Alberta, the Lost River and Wild Horse sites, located in the extreme southeastern portion of the province within the Milk River Watershed (Wershler and Wallis 2002, SRD 2003). Mountain plovers have nested only erratically at these sites and have not established a stable breeding population in the province (Wershler 2000, SRD 2003). Breeding was first confirmed in Alberta in 1979 (Wershler 2000). Breeding has only been recorded in one other location in Canada, in the extreme southwestern corner of Saskatchewan, near Val Marie (SRD 2003). The closest stable breeding population of mountain plovers from Alberta is found in northern Montana, approximately 140 km away (Wershler 2000). A second, core breeding population of mountain plovers is found in Colorado. Collectively, Colorado and Montana are thought to support the majority of the global breeding population of this species (SRD 2003). The mountain plover's breeding range also includes the tablelands of Wyoming, New Mexico, small portions of western Kansas and Oklahoma and the Texas panhandle (Knopf 1996a). Mountain plovers overwinter primarily in California and in parts of northern Mexico, southern Arizona and southern Texas (Wershler 2000).

The significant range-wide decline of mountain plovers in recent decades is thought to have occurred due to a combination of factors: the conversion of vast tracts of native prairie to cropland; habitat loss due to urban and industrial development; current agricultural and range management practices; removal or population reduction of key herbivores such as bison (*Bison bison*), prairie dogs (*Cynomys* spp.) and pronghorn antelope (*Antilocapra americana*); and the suppression of natural fire regimes (Knopf 1996a, Wershler 2000, SRD 2003). Small, isolated breeding populations of mountain plovers in Alberta are also vulnerable to natural events such as weather extremes (*i.e.*, drought or flooding) as well as predation (Wershler 2000).

Over the last 30 years mountain plovers in Canada have been observed most frequently in four areas in Alberta and Saskatchewan. These areas include: Onefour Research Ranch, an 18,000 ha research ranch owned and managed by the Alberta provincial government; Grasslands National Park, owned by the federal Parks Canada Agency; The Wild Horse site, a ranch partly comprised of Crown lease land; and Milk River Natural Area, owned by the Alberta government (COSEWIC 2009).

Two draft Alberta management strategies have been developed for the mountain plover (Wershler 1990, 1991). However, a formal provincial recovery strategy has yet to be implemented. A federal recovery strategy was drafted in the early 1990s (Edwards *et al.* 1993), but was not formalized and implemented until 2006 (Environment Canada 2006).

2.2 Ecology

Mountain plovers are capable of breeding one year after hatching and breed annually thereafter (Graul 1973). They generally arrive on the breeding grounds already paired; otherwise pairing begins immediately after arrival (Knopf 1996a). In Alberta, mountain plovers arrive on the breeding grounds in April and commence breeding in early to mid-May (Wershler 2000). Nesting usually occurs from May to July (Wershler 2000). Mountain plovers display a high degree of breeding site fidelity, usually returning to the same general area to nest (Knopf 1996a, Ellison Manning and White 2001). Returning to an area that is familiar to an individual may improve success in obtaining mates, territory and food (Ellison Manning and White 2001). Mountain plovers will seek out new nesting areas if previously used nesting sites are unsuitable (Wershler 2000).

Male mountain plovers construct small nest scrape depressions on the ground, approximately 9 cm to 10 cm in diameter and approximately 2.5 cm deep (Knopf 1996a). Grass roots, leaves, dried cow manure chips and bits of lichen are often used as nesting material to cover the eggs during incubation (Graul 1975, Knopf 1996a). A typical mountain plover clutch contains three eggs (Graul 1975, Knopf 1996a), although a range of one to four eggs per clutch is possible (Federation of Alberta Naturalists 2007). Under suitable weather conditions and abundant food supplies, mountain plovers may lay two clutches in succession, with the first incubated by the male, and the second by the female (Knopf 1996a, Wershler 2000). Eggs are usually laid 34 to 48 hours apart (Knopf 1996a). Incubation begins once the last egg has been laid (Knopf 1996a). The incubation period is typically 29 days in duration, but can last from 28 to 31 days (Graul 1975, Knopf 1996a).

Mountain plover chicks fledge once they reach 70% of adult body weight, usually 33 to 34 days after hatching (Graul 1975, Miller and Knopf 1993, Knopf 1996a). Adults and fledged chicks usually leave the breeding grounds between late July or mid-August (Knopf 1996a). Most birds arrive at wintering grounds by early November where they gather in small, localized flocks (Knopf 1996a).

2.2.1 Diet

Mountain plovers are opportunistic foragers and display a high degree of dietary flexibility across their range (Knopf 1996a). Winged invertebrates and ground-dwelling invertebrates are the main food sources for this species (Knopf 1996a). Baldwin (1971) conducted a dry-weight biomass study from the stomachs of 13 mountain plovers in Colorado. The author reported that 99.7% of the mountain plover's diet consisted of invertebrates and the remaining 0.3% was comprised of seeds. Invertebrates consumed represented 90 different taxa, with the most important prey items being beetles (Coleoptera) (60%), grasshoppers and crickets (Orthoptera) (24.5%) and ants (Hymenoptera) (6.6%) (Baldwin 1971).

2.2.2 Predators

Mountain plover eggs and chicks are vulnerable to predation from thirteen-lined ground squirrels (*Spermophilus tridecemlineatus*), swift foxes (*Vulpes velox*), American badgers (*Taxidea taxus*) and coyotes (*Canis latrans*) (Knopf 1996a). Common avian predators include Swainson's hawks (*Buteo swainsoni*), prairie falcons (*Falco mexicanus*), loggerhead shrikes (*Lanius ludovicianus*), ring-billed gulls (*Larus delawarensis*) and black-billed magpies (*Pica pica*) (Miller and Knopf 1993, Knopf and Rupert 1996).

Mountain plovers have developed a number of adaptations to reduce predation, including: cryptic plumage coloration; the ability to produce multiple clutches; predator distraction displays by adults; shell removal from the nest at hatching; and rapid movement of chicks from the nest after hatching (Graul 1975, SRD 2003). Despite these adaptations, predation typically has a significant impact on reducing mountain plover nest and fledging success (Graul 1975, Miller and Knopf 1993, Knopf 1996a). Often, greater than 50% of clutches are lost to predators (Miller and Knopf 1993, Knopf 1996a).

2.3 Habitat Requirements

2.3.1 General

In Alberta, mountain plovers are restricted to the Dry Mixedgrass Subregion of the Grassland Natural Region (NRC 2006). The mountain plover is considered a "habitat specialist", occupying areas of open, flat grassland expanses with short, sparse vegetation and high amounts of bare ground (Knopf and Miller 1994, Knopf 1996a, Dechant *et al.* 2002a, SRD 2003). Heavily grazed areas or areas disturbed by fire or prairie dogs are considered highly suitable mountain plover habitat (Knopf and Miller 1994, Knowles *et al.* 1982, Knowles and Knowles 1984, Tipton *et al.* 2008).

2.3.2 Nesting Habitat

Mountain plover nesting sites typically have short vegetation (less than 10 cm in height), extensive amounts of bare ground or lichen cover (between 30% and 50%) and occur in extensive areas (0.5 km to 1 km in diameter) of flat terrain (less than 5% slope) (Graul 1975, Olson and Edge 1985, Knopf 1996a, Dechant *et al.* 2002a, SRD 2003). Nesting sites are typically found in heavily grazed or recently burned areas (SRD 2003). Cow manure piles are often present near nests and may play a role in concealing nests from predators (Graul 1975, Knopf 1996a).

Mountain plover breeding sites in Canada are found in remnant blocks of mixedgrass prairie, including the Sage Creek – Milk River Canyon – Southwest Pasture Complex in Alberta and Grasslands National Park in Saskatchewan (Wershler 2000). Two basic types of breeding habitats have been documented in Alberta at the Lost River and Wild Horse sites (Wershler 2000). Breeding habitat at the Lost River area consisted of extensive, open grassland (with no or a few scattered silver sagebrush [*Artemisia cana*] shrubs) on sandy loam soils formed over outwash materials (Wershler 2000). Habitat at the Wild Horse breeding site was characterized by discontinuous, open grassland within grassland-sagebrush or lower-lying Solonetzic blowout vegetation (Wershler 2000). Dominant plant species found at nest sites at the Lost River site

included thread-leaved sedge (*Carex filifolia*), Sandberg bluegrass (*Poa secunda*), June grass (*Koeleria macrantha*) and blue grama (*Bouteloua gracilis)* (Wershler and Wallis 1986). Bare ground ranged from 15% to 25% on unburned breeding sites to 45% to 50% on recently burned breeding sites in Alberta (Wershler and Wallis 1986).

In several areas of its range, including Grasslands National Park, Saskatchewan, and Montana, mountain plover breeding sites are located within active black-tailed prairie dog towns (Knowles and Knowles 1984, Knowles *et al.* 1982, Olson and Edge 1985, Olson-Edge and Edge 1987, Wershler 2000). Heavy grazing and soil disturbance associated with the combination of black-tailed prairie dogs and livestock provide suitable breeding habitat for mountain plovers (Dinsmore *et al.* 2005, Tipton *et al.* 2008). Prairie dog towns are used for breeding, rearing of young, feeding and roosting (Knowles *et al.* 1982, Olson and Edge 1985). Prairie dogs provide areas of short vegetation or bare ground as a result of their grazing and mound construction. Horizontal visibility and bare ground is significantly greater inside towns, while grass cover and total plant cover is significantly less than in adjacent areas (Knowles *et al*. 1982). Several characteristics were common amongst prairie dog towns occupied by mountain plovers in Montana: all towns were active; vegetation was short; they were situated on upland level areas; they were moderately to heavily grazed by cattle using a four-year, rest-rotation grazing system; and they were at least 6 ha in size (Olson-Edge and Edge 1987). Within towns, mountain plovers nested near small clumps of forbs or pasture sagewort (*Artemisia frigida*) (Olson and Edge 1985). It is believed that cattle grazing contributes to prairie dog habitat, which in turn contributes to mountain plover habitat (Fagerstone and Ramey 1996). Mountain plover nests in Utah are associated with white-tailed prairie dogs (*Cynomys leucurus*) (Ellison Manning and White 2001). Due to their perceived threat as a competitor with domestic livestock and designation as a "pest" species, prairie dogs have been eradicated from the majority of their historical range in North America (Fagerstone and Ramey 1996). Prairie dogs likely inhabited the Milk River Watershed at one time, but their current range in Canada is restricted to the extreme southwestern portion of Saskatchewan. The Richardson's ground squirrel (*Spermophilus richardsonii*), a common burrowing mammal within the mountain plover's range in Alberta, is considered ecologically similar to prairie dogs; however, the role of ground squirrels in maintaining or creating suitable mountain plover habitat needs to be further investigated (Wershler 2000).

Mountain plovers are also known to use fallow or recently plowed fields for nesting in the southern part of its North American range (Knopf 1996a, Shackford *et al.* 1999). However, most nests in cultivated fields are destroyed if fields are seeded in May (Knopf 1996a, Shackford *et al.* 1999). Shackford *et al.* (1999) reported that of 46 nests found on cultivated fields, 70.9% were destroyed by farm machinery. Cultivated fields are therefore regarded as a reproductive 'sink' rather than 'source' habitat, where mortality exceeds reproduction (Shackford *et al.* 1999).

2.3.3 Brood-rearing Habitat

As described above, mountain plovers in Montana typically stay in or adjacent to prairie dog colonies during the brood-rearing period (Olson-Edge and Edge 1987). In Colorado, mountain plover broods were noted to use areas with taller forbs or man-made structures such as fence posts for shade (Graul 1975). Most broods moved away from nest sites within the first three days

after hatching, and typically remained within 300 m of the nest until they fledged (Graul 1975). In Utah, mountain plover broods used moderately dense, low growing (less than 30 cm) shrubby areas with an open understory (Day 1994).

Chicks are especially vulnerable to predators during the 33 to 34 day pre-fledging period (Sordahl 1991). Older, bigger chicks are more likely to run, whereas younger, smaller chicks are more inclined to hide from predators (Sordahl 1991). Chicks are more likely to run in areas of sparse vegetation, and hide in areas with more cover (Sordahl 1991).

2.3.4 Foraging Habitat

Mountain plovers forage most often within the boundaries of their territory during the breeding season, occasionally foraging in other suitable areas (Graul 1973, Knopf 1996a). Characteristic mountain plover foraging habitat consists of extensive areas of disturbed ground with short, sparse vegetation (less than 2 cm) and interspersed patches of bare ground (Knopf 1996a). Prairie dog towns, heavily grazed or trampled areas, unpaved roads and recently ploughed or fallow fields are examples of disturbed areas used for foraging (Knopf 1996a). Olson (1985) found a higher abundance of insect prey in areas with very short, sparse vegetation within prairie dog towns than in adjacent habitats.

Mountain plovers use a flush-pursuit method of foraging for insects, a method well-suited to open areas with short vegetation (Knopf 1996a). This method involves the birds running short distances (of approximately 1 m), then pausing and surveying for moving insects (Knopf 1996a).

2.3.5 Area Requirements

Territories of three male mountain plovers in Colorado were approximately 16 ha, although overlap did occur (Knopf 1996a). A minimum spacing of 100 m between mountain plover nests was recorded in Wyoming (Parrish *et al.* 1993). These nests were placed in 20 m to 120 m patches of suitable habitat, with the majority of nests found in 20 m to 60 m diameter patches (Parrish *et al.* 1993).

Knopf and Rupert (1996) estimated that the minimum area required for brood-rearing by mountain plovers in Colorado was at least 28 ha. Mountain plovers with broods moved an average of 300 m per day, and on average ranged over an area of 56 ha (Knopf and Rupert 1996). Two to three broods can occur in the same general vicinity; therefore, mountain plovers can raise chicks in broadly overlapping areas (Knopf 1996a). Mountain plover densities ranged from 2.0 birds/km^2 to 4.7 birds/km^2 in Colorado and from 6.8 birds/km^2 to 5.83 birds/km^2 in Montana (Knopf 1996a). In Montana, Knowles and Knowles (1998, cited in Wershler 2000) suggested that the persistence of a mountain plover population depends on the availability of a number of suitable sites, widely spaced over a minimum area of approximately 25 km^2.

Olson-Edge and Edge (1987) suggest that prairie dog towns between 10 ha to 50 ha offer productive mountain plover habitat. Smaller prairie dog towns of 6 ha to 50 ha had higher mountain plover densities than larger towns of 100 ha to 300 ha (Olson-Edge and Edge 1987).

3 MCCOWN'S LONGSPUR

3.1 Background

The McCown's longspur is a sparrow-sized grassland passerine with rust-coloured wing patches, a black head and gray on the nape and back of the neck (Semenchuk 1992). Similar to the mountain plover, the McCown's longspur is primarily associated with short grassland vegetation (With 2010) and is considered a primary endemic species (Knopf 1996a). The range of McCown's longspurs in North America is disjunct and extends from southeastern Alberta and southwestern Saskatchewan south to Montana and parts of Wyoming, Colorado, North Dakota, South Dakota and Nebraska (Dechant *et al.* 2002b, COSEWIC 2006). Two breeding populations exist: (1) Alberta, Saskatchewan, Montana, and parts of North Dakota and South Dakota; and (2) Colorado, Wyoming, Nebraska, and part of South Dakota (With 2010). In Alberta, McCown's longspurs breed exclusively in the Grassland Natural Region, although the species has been sighted as far north as Lesser Slave Lake (Semenchuk 1992, NRC 2006, Federation of Alberta Naturalists 2007).

The McCown's longspur is currently listed as 'May Be At Risk' under the Alberta General status listing (GoA 2016). The species is listed as a 'non-game animal' under the provincial *Wildlife Act*. McCown's longspurs and their eggs and nests are also protected from harm under the *Migratory Birds Convention Act*. The McCown's longspur was assessed by COSEWIC in 2006 and given the status of 'Special Concern', a designation the species currently retains (COSEWIC 2006, 2016). Based on the recommendation by COSEWIC, the McCown's longspur was added to *SARA* as a species of 'Special Concern' in 2009. Internationally, the McCown's longspur is listed as a species of 'Least Concern' by the IUCN (BirdLife International 2012).

No population surveys have been published for McCown's longspurs in Alberta, and Alberta Environment and Parks (AEP) has not published a species at risk status report for the species. Environment Canada published a Management Plan for the species in 2014 (Environment Canada 2014). The Canadian population of McCown's longspurs was estimated to be approximately 370,000 individuals and between 50,000 and 500,000 birds in the mid- to late 2000s (COSEWIC 2006, Environment Canada 2014). The species is estimated to be declining at a rate of more than 10% per year (Environment Canada 2014).

3.2 Ecology

McCown's longspurs return to their breeding grounds in southern Canada and the northern United States in April and migrate south to their wintering grounds in September (Semenchuk 1992, With 1994). McCown's longspur females construct cup-shaped nests in small depressions on the ground surface, such that the top of the nest is flush with the ground (Mickey 1943, With 1994). Nest materials consist primarily of grasses (Mickey 1943). Female McCown's longspurs lay clutches of two to five eggs, with an average of three eggs (Mickey 1943, Felske 1971, Semenchuk 1992, With 1994). Four egg nests are more common later in the season and may be related to more abundant food supplies at that time (Felske 1971). Five egg nests are uncommon (Mickey 1943, With 2010). The female incubates the eggs for 12 days, during which time the male will actively feed the female (Semenchuk 1992). Young McCown's longspurs leave the nest at approximately 10 days of age and begin flying at 14 days (Semenchuk 1992). With

(1994) reported a fledging success rate of 56.6% in Colorado. Nests in Colorado had, on average, one successful offspring produced per nest (With 1994). Nest predation has been identified as the primary factor responsible for reproductive failure in grassland birds in general (Best *et al.* 1997, Koford 1999, Jones *et al*. 2010) and McCown's longspurs in particular (Greer and Anderson 1989, With 1994). With (1994) observed a depredation rate of 52.6% for McCown's longspur nests in Colorado. Felske (1971) observed depredation rates of 18.9% and 22.9% for eggs and 33.3% and 40.3% for nestlings, depending on the year. McCown's longspurs often produce double broods (Mickey 1943, Greer and Anderson 1989).

3.2.1 Diet

McCown's longspurs are ground-foragers, walking along the soil surface in search of seeds and insects (Semenchuk 1992). Studies conducted at Matador, Saskatchewan showed that adult McCown's longspurs consumed primarily seeds, which constituted 70% to 89% and 64% to 74% of male and female diets, respectively, during the breeding season (Maher 1974). The remainder of the adult male diet was comprised of arthropods from nine different taxa (Maher 1974). Ants (Hymenoptera) comprised 5% to 14% of adult female diets during the breeding season, while grasshoppers (Orthoptera) constituted 23% of the adult female diet in August (Maher 1974). Three grasshopper species were consumed by adult McCown's longspurs during the breeding season: club-horned grasshopper (*Aeropedellus clavatus*), clear-winged grasshopper (*Camnula pellucida*) and western clouded grasshopper (*Encoptolophus costalis*) (Maher 1974).

Young McCown's longspurs are fed exclusively insects and never regurgitated food (Mickey 1943). Moths (Lepidoptera) and grasshoppers (Orthoptera) are the preferred food source for McCown's longspur nestlings in Wyoming (Mickey 1943). Grasshoppers were listed as the primary food source of McCown's longspurs in Saskatchewan (Fiske 1971). At his Matador, Saskatchewan study site, Felske (1971) observed that 87% of prey items fed by adult McCown's longspurs to nestlings were grasshoppers, the bulk of which included three species: club-horned grasshopper , red-legged grasshopper (*Melanoplus femurrubrum*) and western clouded grasshopper. Club-horned grasshoppers alone comprised 43.8% of nestling diets, ranging from 76% of diets in late June to 18% of diets in the second week of August. Other important prey items fed to nestlings included beetles (Coleoptera), spiders (Araneida) and butterflies/moths (Lepidoptera). Less significant prey items included insects from the following orders: Hymenoptera (bees, wasps and ants), Hemiptera/Homoptera (true bugs), Diptera (true flies) and Odonata (dragonflies and damselflies) (Felske 1971).

3.2.2 Predators

The thirteen-lined ground squirrel was described as the main predator of McCown's longspur nests in Colorado (With 1994). Thirteen-lined ground squirrels will make repeated trips to the same McCown's longspur nests to consume all eggs present (With 1994). In Saskatchewan, Felske (1971) listed the American crow (*Corvus brachyrhynchos*) as the main predator of eggs and the American crow and northern harrier (*Circus cyaneus*) as the main predators of nestlings. Other known or potential predators of McCown's longspurs include: striped skunks (*Mephitis mephitis*), American badgers, swift foxes, red foxes (*Vulpes vulpes*), coyotes, long-tailed weasels (*Mustela frenata*), Richardson's ground squirrels, deer mice (*Peromyscus maniculatus*), northern grasshopper mice (*Onychomys leucogaster*), short-eared owls (*Asio flammeus*), Swainson's

hawks, loggerhead shrikes and bullsnakes (*Pituophis melanoleucus*) (Mickey 1943, Greer and Anderson 1989, With 1994, 2010). Felske (1971) also noted small mammals as other predators of McCown's longspurs in Saskatchewan.

Parasitism rates by brown-headed cowbirds (*Molothrus ater*) on McCown's longspurs have been little studied. In Saskatchewan, Maher (1973) reported that none of the 74 nests he studied were parasitized, whereas Friedmann (1963) found that two of three nests studied in North Dakota were parasitized.

3.3 Habitat Requirements

3.3.1 General

McCown's longspurs occur in open grassland habitat of the Central and Northern Great Plains, including shortgrass and dry mixedgrass prairie with short, sparse vegetation and low amounts of litter (Felske 1971, With 2010). Typical shortgrass habitat contains a mix of short, warm-season grasses, such as blue grama and buffalograss (*Buchloe dactyloides*) intermixed with prickly-pear cactus (*Opuntia polyacantha*) and lesser amounts of mid-height grasses, including western wheat grass (*Pascopyrum smithii*), needle-and-thread (*Hesperostipa comata*) and red three-awn (*Aristeda purpuea* var. *longisteda*) (With 2010). Rabbitbrush (*Ericameria nauseosa*) and pasture sagewort are also common in shortgrass prairie habitats (With 2010). Grass species present in McCown`s longspur habitat in mixedgrass prairie in Saskatchewan included northern wheat grass (*Elymus lanceolatus*), western wheat grass, June grass and green needle grass (*Nassella viridula*) (Felske 1971).

A habitat model developed by Henderson and Davis (2014) for McCown's longspurs inhabiting native prairie in the Milk River Watershed in southwestern Saskatchewan included four variables: vegetation volume; litter mass; percent bare ground cover; and shrub cover. According to the best-fit model, McCown's longspur probability of occurrence in native prairie increases with lower cover of vegetation, litter and shrubs and higher cover of bare ground. A second habitat model developed for McCown's longspurs in Saskatchewan exploring the effect of natural gas well proximity and density on longspur abundance included five variables: vegetation height; litter depth; grass cover; well distance; and well density (Bogard and Davis 2014). According to this model, McCown's longspurs were found further from wells, in sections with at least one well, in habitat with lower grass cover and shorter vegetation heights and in areas where litter was less than 5 mm deep.

Although they prefer native prairie, McCown's longspurs will also nest in both tame pasture and cropland (Davis *et al.* 1997, McMaster and David 1998). Davis *et al.* (1997) found equal abundances of McCown's longspurs in native prairie and tame pasture in southern Saskatchewan. McMaster and Davis (1998) found higher numbers of McCown's longspurs in cropland than in Permanent Cover Program tame pastures and hay fields.

A habitat model developed by Martin and Forsyth (2003) for McCown's longspur use of cropland included three variables: vegetation height; percent bare ground cover; and stubble

count. McCown's longspurs inhabiting cropland preferred lower vegetation heights, higher percent ground cover and higher stubble counts (density).

3.3.2 Nesting Habitat

McCown's longspur nests are placed on the ground next to cow dung or different types of vegetation, including grass of varying heights, cacti and shrubs, although shrubs are preferred (With 1994). While McCown's longspurs may prefer to nest adjacent to shrubs, With (1994) observed nests situated within 1 m of shrubs were two to three times more likely to be depredated than nests located further away from shrubs. In terms of nest-site selection, McCown's longspurs inhabiting a moderately grazed field appeared to select for microhabitat containing cacti within 1 m of the nest and mid-grasses situated within both 1 m and 8 m of the nest, suggesting that the species prefers higher cover for nesting than what is generally available in a given area. However, nesting success was highest for nests that were placed in short grass compared to either mid-grass, cacti or shrubs (With 1994). In Wyoming, Greer and Anderson (1989) observed that McCown's longspur territories had higher percent bare ground, higher vegetation cover within 5 cm of the ground surface, fewer cow pies, lower lichen cover and lower forb cover than unoccupied territories.

3.3.3 Foraging Habitat

More research is needed to adequately describe McCown's longspur foraging habitat and if it differs from its breeding habitat.

3.3.4 Area Requirements

Male McCown's longspurs establish discrete (*i.e.*, non-overlapping) territories on their breeding grounds and vigorously defend them (With 2010). Exact territory size varies by region, including 0.6 ha in Wyoming (Greer and Anderson 1989), 0.5 ha to 1.0 ha in Saskatchewan (Felske 1971), and 0.93 ha (With 1994) and 1.1 ha to 1.4 ha in Colorado (Wiens 1970, 1971).

4 GRAZING AND GRASSLAND BIRD GROUP 1

There have been few published peer-reviewed studies that have assessed how livestock grazing or selective grazing systems affect mountain plovers and McCown's longspurs in Alberta or across their range in North America. However, based on their ecological requirements and the non-peer-reviewed research that has been conducted, grazing appears to be necessary for maintaining suitable habitat for these species.

4.1 Mountain Plover Response to Grazing

Bison, pronghorn and prairie dogs were the primary herbivores of the Great Plains prior to human settlement (Knopf 1996b). Bison and pronghorn preferentially graze on prairie dog towns, intensifying local grazing and trampling pressure at these sites (Coppock *et al.* 1983, Krueger 1986). Yet it is at these sites, where grazing pressure is intense and surface disturbance is excessive, that mountain plovers flourish (Knopf 1996b). Bison have since been extirpated from their former range across the Great Plains, while prairie dog numbers have been reduced by

98 percent throughout the western Great Plains (Knopf 1996b). Similarly, pronghorn populations have been greatly reduced across much of their range in comparison to their estimated pre-settlement abundance (Yoakum *et al.* 1996).

Domestic livestock have replaced bison grazers as the dominant herbivore on the prairie landscape. Strategically managed, cattle grazing can be used to create the habitat conditions required by mountain plovers (Dechant *et al.* 2002a). However, maintaining habitat for mountain plovers presents a challenge as extensive heavily grazed areas are not typically favoured for either long-term sustainable livestock production, maintained range condition, or as habitat for many wildlife species (Wershler 2000). In the same way that eliminating grazing has negative consequences for mountain plovers, conventional range management practices that encourage uniform cover and discourage heavy grazing can diminish habitat opportunities for mountain plovers (U.S. Fish and Wildlife Service 1999, Wershler 2000). As described by Wershler (2000), in conventional range management, "the extreme of heavy grazing is usually avoided or represented in small, localized areas in the overall habitat" (Wershler 2000, p.25). Range management practices such as planting tall, dense exotic grass species (*e.g.*, crested wheatgrass [*Agropyron cristatum*], smooth brome [*Bromus inermis* ssp. *inermis*] or timothy [*Phleum pratense*]), fire suppression, or irrigation projects may also have negative implications for mountain plovers (U.S. Fish and Wildlife Service 1999). Other effects such as incidental cattle trampling destroying mountain plover nests are considered rare and are likely comparable to historical bison trampling (Knopf 1996a).

Grazing will promote suitable breeding habitat for mountain plovers if it removes the previous season's plant litter by the beginning of the breeding season (Wershler and Wallis 2002); maintains patches (minimum 20 m to 60 m) of short (2 cm to 10 cm) and sparse vegetation and approximately 30% bare ground (Parrish *et al.* 1993); and retains scattered patches of taller vegetation to provide shelter for chicks (Graul 1975, Sordahl 1991). Grazing patterns that mimic the heterogeneity that likely existed under a dynamic bison grazing regime will most likely benefit mountain plovers (Wallis and Wershler 1981). Bison are known to have heavily grazed areas before moving on to exploit other areas as forage resources became depleted (Wuerthner 1998). Areas that were heavily grazed likely received periodic rest due to the transient nature of bison and bison population fluctuations due to climatic conditions and other natural factors (Wuerthner 1998). A constantly "shifting mosaic of grazing pressure" is thought to have resulted from the temporal and spatial variability of bison grazing (Wuerthner 1998, p.379).

Wershler and Wallis (2002) recommend intensive winter or early spring grazing to create and maintain habitat for mountain plovers in Alberta. This type of grazing program is recommended for traditional breeding areas and in surrounding high potential habitats. According to Wershler and Wallis (2002), the majority of mountain plover nesting areas in Alberta have been either winter- or spring-grazed. Locally-prescribed heavy summer grazing has also been suggested to promote habitat for mountain plovers in Alberta (Wallis and Wershler 1981). In Colorado, shortgrass pastures that were heavily grazed in the summer were used by mountain plovers for foraging and nesting (Giezentanner 1970, cited in Dechant *et al.* 2002a).

From a livestock productivity standpoint, intensive winter and early spring grazing may not be well-suited to maintaining cattle weight gains or for sustained range health of dry mixedgrass or

mixedgrass prairie. Crude protein and phosphorus levels are highest in actively growing forages and decline substantially as plants become dormant (Holechek *et al.* 2003). Therefore, supplemental feeding is usually required to sustain livestock that are winter-grazed in mixedgrass and dry mixedgrass prairie. Supplemental feeding practices create problems in terms of spread of exotic grasses in mountain plover habitats (Wershler 1991, Wershler 2000). Similarly, grazing early in the spring, prior to green-up, is not recommended due to lower nutritional quality of forage at this stage. Early spring grazing is also not considered sustainable if energy reserves of plants are depleted as new growth is initiated (Holechek *et al.* 2003).

Light to moderate grazing intensity may be appropriate for maintaining mountain plover habitat in areas with naturally sparse, low vegetation cover, during years of severe drought, or in areas with extensive burrowing mammal activity (*e.g.*, prairie dogs or Richardson's ground squirrels) (Wershler and Wallis 2002). Lower grazing intensities would also be appropriate where prescribed burning is conducted in mountain plover habitats (Wershler 2000).

Pyric herbivory, or patch-burn grazing management, is often advocated as a means of replicating historic fire and bison grazing regimes and promoting habitat heterogeneity to benefit grassland songbirds (Fuhlendorf *et al.* 2006). Augustine and Derner (2015) studied the effects of patch-burn grazing on grassland songbirds in the shortgrass prairie of Colorado. All study sites were continuously grazed from May 15 to October 1, with selected patches also subjected to fall prescribed burns. Burned sites had significantly shorter vegetation than control sites. Of the grassland songbirds studied, mountain plovers were found to be significantly more abundant on recently burned sites compared to unburned sites, suggesting that patch-burn grazing management in the shortgrass prairie is an effective tool for maintaining suitable mountain plover habitat.

Few studies have assessed the importance of pronghorn herbivory or Richardson's ground squirrel activity in addition to livestock grazing for maintaining mountain plover habitat in Alberta (Wershler 2000). The combined effect of cattle grazing in combination with pronghorn browsing and ground squirrel activity may be important for reducing silver sagebrush encroachment and maintaining patches of bare ground for mountain plovers. In Montana, all prairie dog towns used by mountain plovers were also grazed by cattle, while stock pond sites without prairie dogs were not used by mountain plovers (Knowles *et al.* 1982).

4.2 McCown's Longspur Response to Grazing

McCown's longspurs generally prefer sparsely vegetated native prairie, whether naturally occurring or as a result of heavy grazing (Felske 1971, With 2010). Based on a review of published literature, Bock *et al.* (1993) concluded that McCown's longspurs generally responded favourably to livestock grazing. In the northern, foothills and transition regions of the Great Plains, Kantrud and Kologiski (1983) found the highest numbers of McCown's longspurs in native prairie that was heavily grazed compared to moderately or lightly grazed fields, whereas in the Southern Great Plains, longspurs were most abundant in moderately grazed fields. In Colorado, Porter and Ryder (1974) found a higher number of McCown's longspurs on heavily grazed shortgrass prairie fields compared to either moderately or lightly grazed fields. At Matador, Saskatchewan, McCown's longspurs were more abundant in heavily grazed native

prairie fields compared to ungrazed prairie fields and fields that were lightly or moderately grazed (Felske 1971, Maher 1973). In Grasslands National Park in Saskatchewan, McCown's longspurs were most frequent in heavily grazed fields in comparison to ungrazed and either lightly or moderately grazed fields (Sliwinski 2011). Wershler *et al.* (1991) suggest McCown's longspurs prefer moderately and heavily grazed sites on dry, sandy sites in southeastern Alberta.

With (1994) compared nesting and reproductive success in McCown's longspur nests in moderately and heavily grazed fields in Colorado. The author observed higher reproductive success, defined as the percentage of young fledged per number of eggs laid in the nest, for McCown's longspurs nesting in heavily grazed native prairie compared to those nesting in moderately grazed native prairie. The heavily grazed field had lower shrub cover than the moderately grazed field, which With (1994) demonstrated was the main habitat variable influencing McCown's longspur nest depredation.

In the pyric herbivory study discussed above, Augustine and Derner (2015) also examined the effects of patch-burn grazing management on McCown's longspurs. Because McCown's longspurs prefer short, sparse vegetation (With 2010), they were expected to benefit from patch-burn grazing management. However, these treatments resulted in no positive benefits for McCown's longspurs in this Colorado study, with longspur abundance independent of patch-burn grazing (Augustine and Derner 2015).

Livestock grazing can also influence the composition of insect communities in grassland habitat, presumably thereby impacting the suitability of that habitat for McCown's longspurs. However, the effect of livestock grazing on insects in unclear (Fleischner 1994). For example, Fielding and Brusven (1995) examined grasshopper densities in Idaho rangeland during drought conditions. They found that grasshopper densities were higher on rangeland that had been rested for 10 years or more compared to rangeland that was subject to rest-rotational grazing. In Arizona, Jepson-Innes and Bock (1989) found that in the summer, grasshopper density was 3.7 times greater on a site protected from livestock grazing compared to a site grazed by livestock, and in the fall, grasshopper density was 3.8 times greater on a site grazed by livestock compared to an ungrazed site. The reason for the apparent contradiction was that different subfamilies with different food preferences were dominant in different seasons. In Colorado, Welch *et al.* (1991) observed that grasshoppers were significantly more abundant on a lightly grazed site than on a heavily grazed site. Because there was no difference between the same sites 19 years earlier, a long-term effect of livestock grazing was the likely cause. A number of European studies also suggest that high grazing intensity can lead to declines in arthropod richness and abundance (*e.g.*, Morris 1967). However, at Matador, Saskatchewan, McCown's longspurs rely on a species of grasshopper that is most abundant on heavily grazed sites early in the season (Felske 1971).

5 GRAZING SYSTEMS AND GRASSLAND BIRD GROUP 1 HABITAT MANAGEMENT

Table III-7 provides an overview of four pertinent grazing systems and their potential positive and negative implications to mountain plovers and McCown's longspurs and their habitat. A grazing system is a tool used to control the spatial distribution, timing, intensity, and frequency

of livestock grazing (Holechek *et al.* 2003). Applied research is needed to properly assess the effects of various grazing systems on mountain plovers and McCown's longspurs in the Milk River and South Saskatchewan River watersheds.

Table III-7 Grazing Systems and Mountain Plover and McCown's Longspur Habitat Management

Grazing System	Discussion
Continuous (Season-Long) Grazing	
Advantages:	Season-long grazing at moderate to heavy stocking rates may create suitable conditions for mountain plovers by creating repeatedly grazed patches with short vegetation and higher amounts of bare ground (Willms pers. comm.). Prescott and Wagner (1996) and *Prescott et al.* (1993) found that McCown's longspurs preferred continuously grazed native prairie fields in southern Alberta compared to other grazing systems, and the species was fairly common in native prairie fields grazed early in the summer. Similarly, Dale and McKeating (1996) observed higher numbers of McCown's longspurs on continuously grazed native prairie fields than fields being complementary grazed. Strategic placement of salt and water can be used to stimulate heavy grazing near to traditional mountain plover and McCown's longspur breeding habitats. Shifting the placement of salt and water over time can allow for periodic recovery of previously heavily used areas. Heavily grazed areas also stimulates habitat for Richardson's ground squirrels, which help to perpetuate patches of bare ground and reduced vegetation height and cover (Fagerstone and Ramey 1996, Wershler 2000). In comparison to rough fescue prairie, mixedgrass and dry mixedgrass prairie communities are typically more resistant to grazing during the growing season; mixedgrass and dry mixed grass prairies fall within the suspected prehistoric bison summer range (Morgan 1980). Repeated defoliation at a moderate utilization level throughout the growing season has been found not to negatively impact species composition in mixedgrass communities (Biondini *et al.* 1998).
Disadvantages:	Season-long grazing at heavy stocking rates may not be sustainable on a long-term basis in terms of cattle weight gains, vegetation production, range health and topsoil conservation. If cattle are consistently grazed at heavy stocking rates under season-long grazing, vegetation heterogeneity will decrease over time, leading to uniform conditions with few tall vegetation patches to provide shelter for mountain plover and McCown's longspur chicks during the brood-rearing season.

Complementary Grazing	
Advantages:	If tame pasture is available within the grazing operation it can be used to defer early season use on native prairie, allowing for recovery of areas that were previously heavily grazed late in the season. Both mountain plovers and McCown's longspurs will nest in tame pastures (Davis *et al.* 1997).
Disadvantages:	Prescott and Wagner (1996) found that McCown's longspurs infrequently occupied tame pastures grazed in spring (*i.e.*, late April to mid-June) in southern Alberta. Unless breeding areas are heavily grazed late in the previous season, delaying early season grazing of these sites may not maintain suitably short vegetation that is required for nesting. Planting tame pastures near to traditional breeding areas may result in encroachment of tall, exotic graminoids into mountain plover and McCown's longspur habitats. McCown's longspurs and mountain plover generally prefer native prairie to tame pasture; therefore, creating new tame pasture fields by cultivating native prairie is unlikely to benefit the species. McMaster and Davis (1998) found that McCown's longspurs were more abundant in cropland than in tame pastures that were part of the Permanent Cover Program. Dale and McKeating (1996) observed higher numbers of McCown's longspurs on continuously grazed native prairie fields than fields being complementary grazed.
Rotational Grazing – Rest Rotation and Deferred Rotation	
Advantages:	Olson and Edge (1985) reported that mountain plovers nested consistently on upland prairie dog towns that were moderately to heavily grazed by cattle under a four pasture rest-rotation grazing system. A three-field rotational grazing system can be strategically managed to provide consecutive early and late season heavier use in at least one field each year. Fields that receive consecutive early and late season use will have short vegetation and low litter cover during the nesting season. Twice-over grazing (*i.e.*, two rotations through the grazed fields in one season) may be appropriate in a two field switch-back grazing system whereby one field is grazed once early in the season and then again late in the season. This would allow the second field to be rested during the early and late seasons every second year. Given appropriate stocking rates, suitable nesting habitat is likely to be created in the field that receives twice-over use. By allowing for periodic rest, rotational grazing systems can facilitate range recovery and sustained livestock yields. As rotational grazing systems encourage overall better utilization of fields, this may encourage, large areas of suitable habitat for mountain plovers. Based on research in Colorado, Knopf and Rupert (1996) estimated that mountain plovers require at least 28 ha of suitable habitat for successfully raising broods. Suitable foraging habitat for mountain plovers is comprised of extensive areas of short vegetation with interspersed bare ground (Knopf 1996a).

Disadvantages:	Prescott and Wagner (1996) and Prescott *et al.* (1993) observed that McCown's longspurs generally avoided native prairie fields that were grazed after July 15th (deferred grazed) and the species preferred fields that were continuously grazed compared to rotationally grazed fields. As prairie dogs do not occur within the Milk River Watershed in Alberta, it is unclear how effective a four pasture rest-rotation grazing system would be for maintaining mountain plover habitat. In the absence of prairie dogs, fields that receive a full season of rest may not maintain suitable nesting conditions for mountain plovers or McCown's longspurs.
Rotational Grazing – High Intensity Grazing	
Advantages:	As mountain plover and McCown's longspur nest sites are primarily associated with heavily grazed habitats, large areas of intensive grazing is recommended for maintaining habitat for these species (Knowles *et al.* 1982, Knopf 1996a, Wershler 2000, Dechant *et al.* 2002a). Wershler and Wallis (2002) recommend intensive winter or spring grazing in traditional breeding areas and in surrounding high potential habitats. Heavy grazing over large areas may also promote conditions that are suitable for Richardson's ground squirrels, an important prey species for numerous other wildlife species. Richardson's ground squirrels may also help to perpetuate suitable habitat for both mountain plovers (Wershler 2000) and McCown's longspur.
Disadvantages:	Intensive, heavy grazing may have negative long-term consequences for livestock and rangeland productivity if it leads to a significant decline in range health. Intensive grazing over large areas can negatively affect habitat for other wildlife species that require greater amounts of cover for nesting, shelter or foraging. Intensive grazing usually results in rapid deterioration of riparian areas in particular, with potential negative consequences for numerous wildlife species that rely on these areas (Fitch and Adams 1998).

6 BENEFICIAL MANAGEMENT PRACTICE RECOMMENDATIONS

The following general land-use and grazing recommendations offer a variety of means to protect or enhance mountain plover and McCown's longspur habitat within the Milk River and South Saskatchewan River watersheds. Further research is required to improve our understanding of these species and their habitat requirements as well as their response to various grazing management practices (see Section 7).

Critical for mountain plovers and McCown's longspurs is the ongoing maintenance of extensive tracts of dry mixedgrass or mixedgrass prairie with substantial areas of short grass (2 cm to 10 cm), few shrubs, low litter cover and interspersed patches of bare ground (Wallis and Wershler 1981, Henderson and Davis 2014). Controlled grazing or prescribed burning is necessary for maintaining these habitat characteristics. As mountain plovers repeatedly use the same areas for

breeding, management efforts should focus on protecting and maintaining suitable habitat at traditional breeding grounds in the Lost River and Wild Horse areas (SRD 2003). Surrounding areas with high potential habitat should also be appropriately managed to enhance breeding opportunities for mountain plovers (Wershler 2000).

Although there is a low possibility that mountain plovers will establish a self-sustaining population in Alberta, maintaining small, northern breeding populations contributes to the overall diversity of the species, particularly since adjoining populations in northern Montana have been extirpated (Wershler 2000). Protecting mountain plover habitat may also afford protection to several other rare fauna and flora that occur within the plover range in Alberta (Wershler 2000).

6.1 General Recommendations

Native Prairie Conservation and Restoration

- Within known mountain plover and McCown's longspur breeding areas, protect remaining dry mixedgrass and mixedgrass native prairie habitat from cultivation. Although both species will nest in tame pasture and cropland, native prairie is generally preferred (Felske 1971, Davis *et al.* 1997, McMaster and Davis 1998, With 2010). Loss and degradation of native prairie habitat have been described as the main threats to McCown's longspurs in the United States (Sedgwick 2004). Cultivated fields are regarded as sink habitat for mountain plovers (Shackford *et al.* 1999).
- Develop conservation agreements with private landowners that focus on conservation of native prairie at key sites (Environment Canada 2014).
- Remove marginal cropland from production, where possible, and seed with native mixedgrass or dry mixedgrass graminoids to enlarge suitable habitat for mountain plovers and McCown's longspurs. When seeding to native species, work with a knowledgeable Agrologist, Biologist or Reclamation Specialist to ensure selection of appropriate native plant species for the area being restored/reclaimed as well as appropriate seeding rates.
- Reclamation of native grassland should follow the provincial guidelines for native prairie reclamation (ESRD 2013). Natural recovery is preferable to seeding for native prairie community reclamation following development, provided that erosion risks and invasion by non-native species are of low concern (Hickman 2010). Preferred native species and seeding rates should be specified in industrial development application documents if a company chooses against natural recovery (Hickman 2010). Certified weed-free native or wild harvested native seed should be required when seeding in intact silver sagebrush communities (Hickman 2010).

Habitat Management

- Consider planting winter wheat in areas where croplands exist within the vicinity of known mountain plover breeding areas (Shackford *et al.* 1999). Winter wheat fields require fewer disturbances in spring and early summer and may permit mountain plovers to successfully breed in cultivated fields (Shackford *et al.* 1999). Winter wheat is usually seeded into standing stubble in late August or early September, and harvested early the following August.

- Promote the use of minimum tillage practices in cropland with known breeding pairs of McCown's longspurs. Martin and Forsyth (2003) found that while McCown's longspurs nesting in cropland occurred more frequently in conventional tillage versus minimum tillage, the species was more productive under a minimum tillage regime. Minimum tillage also benefited a number of other grassland songbirds compared to conventional tillage. In terms of cover type, McCown's longspurs were more common and productive in summer fallow and spring cereal crops compared to winter wheat (Martin and Forsyth 2003).
- Avoid plowing during the nesting season in cultivated fields that may be used for breeding by mountain plovers and McCown's longspurs (Knopf 1996a). Nests, eggs and chicks in cultivated fields are at risk of being destroyed by farm machinery if planting occurs during the nesting period (*i.e.*, May to June) (SRD 2003). Shackford *et al.* (1999) noted that of those nests that failed in cultivated fields, more than 70% failed as a result of destruction by farm machinery.
- Avoid developing irrigation systems on marginal land within suitable mountain plover and McCown's longspur habitats.
- Maintain grazing as a disturbance process within known mountain plover and McCown's longspur breeding habitats. Mountain plovers and McCown's longspurs prefer nesting in heavily grazed areas with short vegetation. Livestock grazing can therefore be used as a tool to maintain optimal vegetation characteristics in critical mountain plover and McCown's longspur habitats (see Grazing Recommendations, below).
- Consider the use of prescribed burning at known mountain plover and McCown's longspur breeding sites to maintain suitable habitat, particularly in areas where grazing disturbance has been removed (Wershler 1991, Knopf 1996a, Dechant *et al.* 2002a,b). Prescribed burning can be used to create and maintain areas of shorter grass within mixedgrass prairie (Dechant *et al.* 2002a). Burns should be carefully conducted in late summer or early fall (during appropriate conditions) to promote suitable habitat for the next breeding season (Wershler 1991, Dechant *et al.* 2002a). The main plant species in mountain plover nesting habitats are most fire-resistant during late summer or early fall, and this is also the season when natural lightning strike fires occur in the region (Wershler 1991). Prescribed burns should only be conducted under the supervision of experienced professionals.
- Maintain populations of burrowing small mammals such as Richardson's ground squirrels that may serve a similar ecological function as black-tailed prairie dogs (SRD 2003). Ground squirrels maintain areas with short vegetation and their burrowing activities increase bare soil cover. Discontinue Richardson's ground squirrel control on Crown lands in the Lost River region (Wershler 1991). Promote natural predators as biological rodent control agents. Promote dual use of mountain plover habitats for pronghorn by ensuring that barbed wire fences allow for pronghorn passage (Yoakum *et al.* 1996). Paige (2008) recommends the following adjustments for wildlife friendly fences: the lower two strands of barbed wire should be replaced with smooth wire. In addition, the bottom wire should be moved up to a height of 40 cm to 45 cm to allow pronghorn to pass underneath. A minimum of 30 cm of space should be present between the top two wires. The top wire should be no more than 90 cm high

- Minimize the spread of exotic plants in mountain plover and McCown's longspur habitats (Wershler 1991, Dechant *et al.* 2002a). Tall exotic grasses such as crested wheat grass, smooth brome and timothy are not adapted to tolerate intense grazing and their dense, tall growth habit does not create suitable habitat for mountain plovers and McCown's longspurs (SRD 2003, Sedgwick 2004). Companies should be prohibited from seeding crested wheat grass and other invasive grass species during reclamation/restoration activities (Hickman 2010). Reclaim pipeline rights-of-way or roadside ditches using native species or allow for natural recovery, where appropriate.
- Ensure mountain plover and McCown's longspur habitat needs are considered in any new or updated management plan for public land within the ranges of the two species (Environment Canada 2014).

Habitat Protection and Industrial Development Mitigation

- Protect the known Lost River and Wild Horse mountain plover breeding sites in Alberta from industrial development or fragmentation by linear disturbances (*e.g.*, road construction) (Wershler 1991, U.S. Fish and Wildlife Service 1999, Wershler 2000). Protecting traditional breeding sites is important as mountain plovers usually return to the same sites to breed year after year (Knopf 1996a).
- Abide by Alberta Environment and Parks (AEP) recommended set-back distances and timing restrictions for mountain plover nests in the Grassland Natural Region (GoA 2011). Mountain plover nests and surrounding habitat have a 100 m set-back from all levels of disturbances (*i.e.*, low, medium and high) from April 1st to July 15th. All potential disturbances, from oil and gas development to recreational and agricultural activities, should abide by these timing and set-back recommendations. Adult mountain plovers have been known to abandon eggs after being disturbed on the nest and stress-related mortality has also been reported among adults (Graul 1975). Human activity can also lead to intolerable levels of exposure of chicks to natural elements (*e.g.*, temperature extremes) (U.S. Fish and Wildlife Service 1999). Graul (1975) found that mountain plover chicks less than two weeks old died from heat exposure within 15 minutes without shade when temperatures exceeded 27°C (Graul 1975).
- Consider developing and implementing timing and set-back restrictions for McCown's longspur nests in the near future. McCown's longspurs in Saskatchewan showed a complicated relationship to gas development, being found in higher numbers at greater distances from gas wells but also being found in greater numbers in sections with at least one gas well compared to sections with no gas wells (Bogard and Davis 2014).
- Conduct pre-development wildlife surveys to locate mountain plovers and McCown's longspurs or their nests in areas with suitable habitat or known populations. Re-route planned developments where necessary. Ensure wildlife survey data is entered into the Fish and Wildlife Management Information System (FWMIS) database maintained by AEP. Consider also reporting 'zero data' in areas that have been searched for species at risk but in which none were found.
- Ensure that proposed developments that are subject to the environmental assessment process consider the needs of McCown's longspurs and mountain plovers (Environment Canada 2014).

- Use Protective Notations under the *Public Lands Act* to protect key mountain plover and McCown's longspur habitat.
- Enforce provisions under the *Wildlife Act* and *Migratory Birds Convention Act* protecting mountain plovers and McCown's longspur nests from destruction.

Pest Control

- Avoid the use of pesticides, including insecticides used to control grasshoppers, in the vicinity of suitable mountain plover and McCown's longspur breeding habitats (Wershler 2000, SRD 2003, Environment Canada 2006). Although more research is needed to assess the effects of pesticides on songbird survival or reproductive success, mountain plovers and McCown's longspurs are susceptible to exposure to pesticide residue as they are dependent on insects for food for nearly all or a portion of their life cycle. Preliminary studies have found that levels of DDE and selenium may be harmful to mountain plovers (U.S. Fish and Wildlife Service 1999). The use of pesticides may also result in diminished prey availability, which may affect mountain plover productivity (U.S. Fish and Wildlife Service 1999). Mineau *et al.* (2005) found correlations between pesticide use and declines of grassland birds in prairie farmland.

Road Mortality

- Limit vehicular traffic through mountain plover breeding habitats during the breeding season (Wershler 2000, Environment Canada 2006). Mountain plovers often nest or feed near roads or use them as travel corridors and may be killed or injured by vehicles (U.S. Fish and Wildlife Service 1999, Wershler 2000).

Public Education and Awareness

- Develop a public awareness campaign for landowners, stakeholders and visitors to the Lost River and Wild Horse areas (Wershler 1991).
- Brochures, posters or information signposts can be used to inform the public about the habitat requirements of mountain plovers and McCown's longspurs. Public awareness campaigns are important for promoting voluntary participation in population monitoring programs and promoting adoption of beneficial management practices.
- Offer incentives to landowners for maintaining species at risk populations on their land. Provide financial or other compensation to landowners that undertake habitat enhancements. Offer public acknowledgement and appreciation to landowners who undertake initiatives to benefit species at risk.
- Establish and maintain good relationships with landowners in southern Alberta whose land contains species at risk. Be willing to listen to the concerns of landowners.
- Disseminate knowledge of habitat requirement and BMPs for mountain plovers and McCown's longspurs among land managers, landowners and crown leaseholders.
- Work with farmers that have mountain plovers and/or McCown's longspurs nesting on their land to avoid seeding and cultivating around nests until after eggs have hatched (Environment Canada 2006).

Population Monitoring

- Conduct consistent annual surveys of mountain plovers in high potential habitats and in traditional breeding areas in southern Alberta (Wershler and Wallis 2002, SRD 2003). Surveys should be conducted in mid-April to early May when birds are returning from wintering grounds as well as during the nesting phase (early May to mid-June) (Wershler and Wallis 2002). Conduct surveys across a wider range of habitats once every five years (Environment Canada 2006).
- Continue to document population sizes of nesting mountain plovers, habitat use and nesting success (Wershler and Wallis 2002). Collect additional information on mountain plover habitat use before and after the breeding season (Wershler 1991).

6.2 Grazing Recommendations

Maintaining grazing disturbance is important for continued provision of suitable habitat for mountain plovers and McCown's longspurs (Knopf 1996a, Wershler 2000, Dechant *et al.* 2002a,b). As the extremely heavily grazed conditions favoured by mountain plovers and McCown's longspurs are not appropriate for many other wildlife species, controlled or strategic grazing should be used to selectively manage for mountain plover/McCown's longspur habitat rather than promoting extensive heavy grazing in the region (Payne and Bryant 1994). A range management program for mountain plovers and McCown's longspurs should be restricted to key areas of known habitat as well as areas of surrounding high potential habitat (Wershler and Wallis 2002). Appropriate grazing systems should be developed site-specifically for ranches participating in conservation program in southern Alberta.

Grazing management geared toward mountain plovers and McCown's longspurs should aim to perform the following functions:

- Maintain sufficient patches of short grass and bare ground for foraging and nesting (Knopf 1996a).
- Remove previous season's litter build up prior to the start of the breeding season (Wershler and Wallis 2002).
- Limit shrub cover (Henderson and David 2014).
- Control the spread of tall, exotic grasses in nesting areas (Wershler 1991).
- Promote insect species abundance and diversity.
- Promote habitat for burrowing small mammals that are ecologically similar to prairie dogs (Wershler 1991, 2000).

To perform these functions, the following range management principles should be applied:

- Create and maintain large (minimum 20 m to 60 m diameter) patches of heavily grazed vegetation (2 cm to 10 cm) and approximately 30% bare ground in known mountain plover breeding areas (Parrish *et al.* 1993, Knopf 1996a).
- A safe use factor of 25% to 50% is generally recommended for native grassland in the Dry Mixedgrass and Mixedgrass Natural Subregions (Adams *et al.* 2013a,b). However, this degree of utilization may not be appropriate for maintaining mountain plover and

McCown's longspur habitat. In the case of these species, increased utilization may be warranted based on local environmental conditions and an initial assessment of the effectiveness of distribution tactics or timing of grazing to create suitable conditions for mountain plovers.

- Avoid uniform grazing of grasslands (Wallis and Wershler 1981). Habitat heterogeneity on a landscape scale is recommended to provide habitat for the greatest number of grassland songbirds (Fuhlendorf *et al.* 2006, Sliwinski and Koper 2015).
- Apply controlled, late-season or early-season heavy grazing near to traditional mountain plover and McCown's longspur breeding areas. This may be accomplished by either three-field rotational grazing whereby at least one field receives consecutive early and late season use; or by a two-field system using twice-over grazing (where a field is grazed twice during one season).
- Season-long grazing at a suitable stocking rate may be appropriate for meeting mountain plover and McCown's longspur habitat needs if sufficiently large patches of heavy use are perpetuated (Prescott and Wagner 1996, Willms pers. comm.)
- Strategically distribute salt and water to encourage heavier use near preferred mountain plover and McCown's longspur nesting areas (Wershler 2000).
- Vary intensity and frequency of use in mountain plover breeding areas in accordance with fluctuations in precipitation and local environmental conditions. In other words, less use may be required during drought years or in areas with naturally sparse vegetation, while twice-over late-season grazing may be necessary in years of higher precipitation or in more productive range sites.
- Reduce grazing intensity in mountain plover habitat in recently burned areas (Wershler 2000).
- Retain some patches of taller vegetation in brood rearing areas (a 28 ha area around nest sites (Knopf and Rupert 1996) to provide shade and predator escape shelter for chicks (Graul 1975, Sordahl 1991).
- Control the spread of exotic invasive graminoids in mountain plover nesting habitat due to supplemental feeding practices. Avoid placing feeding stations near to existing or high potential nesting areas. Where possible, limit feeding stations to existing cultivated areas or tame pastures to avoid exotic species encroachment in native prairies (Wershler 1991).
- Avoid creating new tame pastures near traditional mountain plover and McCown's longspur habitats.
- Avoid creating access roads, corrals or ranch buildings within traditional mountain plover and McCown's longspur breeding areas or in surrounding high potential habitats (Wershler 2000).

7 RESEARCH RECOMMENDATIONS

Applied research is warranted to better assess management practices that will maintain suitable mountain plover and McCown's longspur habitat in the Milk River and South Saskatchewan River watersheds. Ongoing monitoring of mountain plover and McCown's longspur populations in Alberta is important to assess population trends and response to habitat management initiatives (SRD 2003).

Key research needs are described below for each species.

7.1 Mountain Plover

- Evaluate the effectiveness of different livestock grazing practices (*i.e.*, grazing intensity, grazing system and timing of grazing) for creating or maintaining suitable mountain plover habitat (Knopf 1996a). Conduct grazing research in potential nesting habitats adjacent to traditional breeding sites (Wershler 1991). Attempt to determine which grazing strategies and stocking rates are best suited for maintaining sufficient habitat for mountain plovers while also supporting sustained livestock and rangeland productivity. Develop recommended stocking rates (Animal Unit Months [AUMs]/acre) for maintaining mountain plover habitat. Research previously conducted at the Onefour research station should help inform recommended grazing practices (Willms pers. comm.).
- Investigate the use of prescribed burning alone and with various combinations of grazing as a tool to create or maintain mountain plover habitat (Wershler 1991).
- Evaluate mountain plover nesting success in relation to various management techniques (*e.g.*, grazing and/or prescribed burning) and habitat characteristics (*e.g.*, vegetation structure, plant species composition and amount of bare ground).
- Investigate the contribution of pronghorn herbivory in combination with cattle grazing for maintaining suitable mountain plover habitat.
- Investigate the importance of Richardson's ground squirrels for maintaining suitable breeding habitats for mountain plovers at the Lost River and Wild Horse sites (Wershler 2000).
- Investigate methods to reduce shrub encroachment in mountain plover breeding habitats where shrub growth exceeds desired thresholds for suitable nesting habitat.
- Develop a Resource Selection Function (RSF) model for mountain plover probability of use for habitats in southeastern Alberta. The resulting Geographic Information System (GIS) maps should be field verified and may prove useful in identifying key mountain plover habitat.
- Research nesting success of mountain plovers nesting in native prairie versus tame pasture and cropland in southeastern Alberta.

7.2 McCown's Longspur

- Determine the population size and trend of McCown's longspurs in Alberta using statistically rigorous survey methods. The Canadian Wildlife Service has begun collecting and compiling observational data for McCown's longspurs in Alberta and Saskatchewan (Environment Canada 2014). The data is being organized in spatial format and will be used to develop an up-to-date distribution map for the species in Canada. Additional monitoring of McCown's longspurs is being conducted as part of the Breeding Bird Survey, Grassland Bird Monitoring program and Christmas Bird Count (Environment Canada 2014).
- Evaluate McCown's longspur nesting success in relation to various management techniques (*e.g.*, grazing and/or prescribed burning), habitat characteristics (*e.g.*,

vegetation structure, plant species composition and amount of bare ground) and land-use practices (*e.g.*, native prairie versus cropland versus tame pasture).

- Develop an RSF model for McCown's longspur probability of use for habitats in southeastern Alberta. The resulting GIS maps should be field verified and may prove useful in identifying key longspur habitat.
- Further define preferred habitat characteristics for McCown's longspurs in the Dry Mixedgrass Natural Subregion of Alberta (NRC 2006).
- Evaluate the effectiveness of different livestock grazing practices (*i.e.*, grazing intensity, grazing system and timing of grazing) for creating or maintaining suitable McCown's longspur habitat. Develop recommended stocking rates (Animal Unit Months [AUMs]/acre) for maintaining McCown's longspur habitat.
- Investigate the use of prescribed burning alone and with various combinations of grazing as a tool to create or maintain McCown's longspur habitat.
- Better determine anthropogenic threats to McCown's longspur survival and persistence on breeding and wintering grounds (Environment Canada 2014).
- Research appropriate set-back distances for McCown's longspurs from industrial development.
- Research the extent of natal philopatry and adult site fidelity and dispersal (Sedgwick 2004).

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F. GRASSLAND BIRD GROUP 2 (LONG-BILLED CURLEW, UPLAND SANDPIPER, SPRAGUE'S PIPIT, BAIRD'S SPARROW, GRASSHOPPER SPARROW AND CHESTNUT-COLLARED LONGSPUR)

The purpose of this report is to summarize and compare the ecology and habitat requirements of six grassland bird species found within the Milk River and South Saskatchewan River watersheds: long-billed curlew (*Numenius americanus*), upland sandpiper (*Bartramia longicauda*), Sprague's pipit (*Anthus spragueii*), Baird's sparrow (*Ammodramus bairdii*), grasshopper sparrow (*A. savannarum*) and chestnut-collared longspur (*Calcarius ornatus*). Based on the ecology and habitat requirements of these grassland birds, the potential effects of grazing and various grazing systems on these grassland birds and their habitats are discussed. This discussion is followed by a summary of recommended beneficial management practices (BMPs) to enhance grassland bird habitat in the Milk River and South Saskatchewan River watersheds in Alberta. These recommendations can be applied to the range of these species within the Grassland Natural Region of Alberta (Natural Regions Committee [NRC] 2006). Lastly, a brief summary of additional research needs is presented.

Long-billed curlews, Sprague's pipits, Baird's sparrows and chestnut-collared longspurs are considered primary endemic species of the Great Plains, having evolved and occurring exclusively on the prairies of North America (Knopf 1996a). Upland sandpipers and grasshopper sparrows are considered secondary or more widespread species, but have a strong affinity to the Great Plains (Knopf 1996a). Due in large part to the large-scale loss, fragmentation and alteration of native prairie, endemic grassland birds as a group have reportedly "shown steeper, more consistent, and more geographically widespread declines than any other behavioural or ecological group of North American species" (Knopf 1996a, p.147).

Presently, all six grassland bird species discussed in this section, as well as their eggs and nests, are protected from hunting or collection in Canada under the federal *Migratory Birds Convention Act*. All six of these species are further protected as 'non-game animals' under Alberta's *Wildlife Act*. Despite these forms of protection, grassland birds continue to be threatened by habitat loss and/or agriculture. Native prairie offers the greatest potential to retain habitat for grassland birds, particularly if livestock grazing can be used to enhance or maintain suitable habitat. Historically, bison grazing on the Great Plains played a major role in stimulating variation in vegetation structure which in turn created habitat for a diversity of bird species.

1 LONG-BILLED CURLEW

1.1 Background

The long-billed curlew, North America's largest shorebird, breeds preferentially in mixedgrass and fescue prairie and sandhills in southern Alberta (Prescott and Bilyk 1996, Saunders 2001). The species generic name, *Numenius,* translates as "crescent moon" and refers to the long-billed curlew's long, down-curved bill (Dugger and Dugger 2002). In addition to its long bill, this species' buffy colouration and long legs make this bird distinctive (Semenchuk 1992). Within Canada, long-billed curlews breed predominantly in the Grassland Natural Region of southern

Alberta (NRC 2006), in south-central and southwestern Saskatchewan and in the dry grasslands of southern interior British Columbia (Hill 1998). In the United States, long-billed curlews breed in shortgrass and mixedgrass habitats of the Great Plains, Great Basin and intermontane valleys of 16 states in the northwestern, central and south-central regions (Hill 1998, Dugger and Dugger 2002). The largest remaining populations of long-billed curlews occur in the Great Plains of Montana (Hill 1998). The long-billed curlew's winter range includes coastal and inland habitats in the southern United States (primarily in California, Texas and Louisiana) and extends into Mexico and Central America (Hill 1998, Dugger and Dugger 2002).

The long-billed curlew is currently listed as 'Sensitive' in Alberta at the General Status level (Government of Alberta [GoA] 2016). The long-billed curlew has been designated as a species of 'Special Concern' under the provincial *Wildlife Act*. Nationally, the long-billed curlew was designated as a species of 'Special Concern' by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) in 1992 (De Smet 1992a). The species was re-examined in 2002 and the status of 'Special Concern' was retained (COSEWIC 2002). The long-billed curlew remains a species of 'Special Concern' nationally (COSEWIC 2016). The species was listed under Schedule 1 of the *Species at Risk Act* in 2005. Internationally, the long-billed curlew is listed as a species of 'Least Concern' by the International Union for Conservation of Nature and Natural Resources (IUCN) (BirdLife International 2012a).

Formerly abundant throughout most of the prairie regions in Canada and the United States, long-billed curlew populations have declined substantially throughout their breeding and wintering range since the early 1900s (Hill 1998, COSEWIC 2002a). Long-billed curlews have been extirpated from Kansas, Michigan, Iowa, Minnesota, Wisconsin, eastern Nebraska, Illinois, Manitoba and southeastern Saskatchewan (Hill 1998). Although a substantial change in population size has not been noted in Canada over the last 10 years, there have been regional declines and the species is still thought to be declining (Environment Canada 2013, GoA 2016). The minimum population estimate of long-billed curlews in Canada is 43,000 adults (Environment Canada 2013).

Over-hunting in the 1800s and early 1900s and elimination of breeding habitat are thought to be primarily responsible for long-billed curlew declines, particularly in the eastern half of their historic range (Dugger and Dugger 2002). Loss and degradation of prairie breeding habitat is presently considered among the greatest threats to the stability of long-billed curlew populations (Renaud 1980, De Smet 1992a, Hill 1998, Saunders 2001, Dugger and Dugger 2002). Agriculture and industrial and urban development are dominant causes of habitat loss (Dugger and Dugger 2002). Although not as significant as habitat loss, the use of organochlorine pesticides has caused documented mortalities and other sublethal effects in long-billed curlews (Blus *et al*. 1985).

1.2 Ecology

Long-billed curlews arrive on their breeding grounds in southern Alberta in mid- to late April (Hill 1998, Saunders 2001). Nest building usually begins in early May, and the nesting period typically lasts until mid-June, with brood-rearing occurring from June to July and possibly into early August (Renaud 1980, Hill 1998). Long-billed curlews are extremely territorial during the

breeding season, vigorously defending a nesting territory from pre-laying through to chick-hatching (Dugger and Dugger 2002).

The long-billed curlew is a long-lived, late-maturing species with a low reproductive output (Hill 1998). The average age at first breeding is three to four years for males and two to three years for females (Redmond and Jenni 1986). On average, long-billed curlews have an eight to 10 year life-span (Redmond and Jenni 1986). Females usually lay one clutch each breeding season, with clutch sizes averaging four eggs and ranging from two to five eggs (Dugger and Dugger 2002). Re-nesting following nest failure is considered rare (Redmond and Jenni 1986). A full clutch takes four to seven days to complete, with eggs laid on alternate days (Hill 1998). Both females and males will incubate the eggs; females will sit during the day and males at night (De Smet 1992a). The incubation period generally lasts 27 to 30 days (Dugger and Dugger 2002). Long-billed curlew chicks will hatch synchronously, often within a period of 5 hours (Hill 1998). Chicks are precocial, meaning they are capable of walking and feeding themselves shortly after hatching (Dugger and Dugger 2002). Broods are actively defended, especially by the male (Dugger and Dugger 2002). Adult females often abandon broods at two to three weeks after hatching, after which time the male will care for the young until they fledge, approximately 41 to 45 days after hatching (Hill 1998). Male long-billed curlews are often more likely to return to a breeding area than females (Redmond and Jenni 1982). If a female loses a clutch or is exposed to excessive disturbance during nesting, she is less likely to return to the same nesting area in subsequent years than successful females (Redmond and Jenni 1982).

Long-billed curlews form post-breeding flocks in July and August in preparation for the fall migration (Hill 1998). Most long-billed curlews leave Alberta to begin migration by the end of August (Hill 1998).

1.2.1 Diet

Long-billed curlews are considered opportunistic foragers on breeding grounds, foraging primarily on grasshoppers and beetles, and occasionally on small vertebrates such as bird eggs and nestlings (horned larks [*Eremophila alpestris*] in particular) (Sadler and Maher 1976, Redmond and Jenni 1985, Dugger and Dugger 2002). A study in western Idaho investigated the diet of long-billed curlew chicks (Redmond and Jenni 1985). Five insect orders and one arachnid order were identified from the stomach contents of nine chicks; grasshoppers and carabid beetles dominated the diet samples (Redmond and Jenni 1985).

The long, downward-curved bill of the long-billed curlew is thought to be an adaptation for foraging for earthworms or burrowing species, such as crab and shrimp in the tidal mudflats of their winter ranges (Dugger and Dugger 2002). When long-billed curlews forage on their breeding grounds they use a pecking method, whereas in their winter ranges they tend to use a probing method (Dugger and Dugger 2002).

1.2.2 Predators

As a ground-nesting species, long-billed curlew eggs, nests and chicks are prone to predation by avian and mammalian predators (Redmond and Jenni 1986, Pampush and Anthony 1993, Dugger and Dugger 2002). A study in western Idaho found that 42% of all long-billed curlew clutches

failed, with predators responsible for the majority (*i.e.*, 84%) of nest failures (Redmond and Jenni 1986). Hartman and Oring (2009) observed a mean nest success rate of 31% (or 45% if second broods were considered) in Nevada in a landscape dominated by hay fields, with predation as the leading cause of nest failure and chick mortality.

Redmond and Jenni (1986) found that American badgers (*Taxidea taxis*) and canids, including red fox (*Vulpes vulpes*), coyotes (*Canis latrans*) and feral dogs (*C. familiaris*), were the most significant predators of long-billed curlew eggs and clutches in Idaho. Similarly, Hartman and Oring (2009) found that mammalian predators (*e.g.*, coyotes) were the main nest predators of long-billed curlews in Nevada. Other common predators of long-billed curlew eggs include black-billed magpies (*Pica pica*), common ravens (*Corvus corax*), American crows (*Corvus brachyrhynchos*) and bullsnakes *(Pituophis melanoleucus*) (Allen 1980, Redmond and Jenni 1986, Pampush and Anthony 1993). Predators that target long-billed curlew chicks include Swainson's hawks (*Buteo swainsoni*), ferruginous hawks (*B. regalis*), great horned owls (*Bubo virginianus*), black-billed magpies, prairie falcons (*Falco mexicanus*) and long-tailed weasels (*Mustela frenata*) (Allen 1980, Redmond and Jenni 1986, Hartman and Oring 2009).

Long-billed curlews rely on the cryptic colouring of their plumage and eggs to evade predators (Redmond 1986). Long-billed curlews will also clump their breeding territories in loose aggregations to aid in predator defense (Hill 1998, Saunders 2001).

1.3 Habitat Requirements

1.3.1 General

The long-billed curlew is considered to be endemic to the Great Plains of North America (Dugger and Dugger 2002). Long-billed curlews typically occur in open expanses of level to gently rolling shortgrass and mixedgrass native prairie habitats during the breeding season (Renaud 1980, Hill 1998, Dechant *et al.* 2002a). In southern Alberta, there is a positive relationship between long-billed curlews and native prairie during the breeding season, with long-billed curlews twice as numerous in areas comprised of more than 50% native prairie than in areas containing lesser amounts of native prairie (*i.e.*, 0% to 50%) (Saunders 2001). Although long-billed curlews showed a preference for native prairie, 36% of curlew observations were in cultivated areas (Saunders 2001). Saunders (2001) also reported a negative correlation between long-billed curlews and riparian areas. Similarly, Gratto-Trevor (1999) found that long-billed curlew nests near Brooks, Alberta were greater than 1 km from permanent waterbodies, including dugouts. This contrasts with research in Colorado, where the majority of long-billed curlew observations were within 400 m of water (McCallum *et al.* 1977). Owens and Myres (1973) found that long-billed curlews were 4.5 times more abundant in native prairie than in cultivated lands near Hand Hills, Alberta. In the Southern Prairie Biome of Alberta, which encompasses grasslands south of the South Saskatchewan and Oldman rivers and east of the Rocky Mountain foothills, Prescott and Bilyk (1996) observed long-billed curlews in 16.7% of sandhill (n=12) and fescue sites (n=18); 22.2% of native mixedgrass sites (n=27); and 6.3% of tame pasture sites (n=16). Long-billed curlews were similar in abundance in the fescue, mixedgrass and sandhill sites, and were half as numerous in tame pasture sites (Prescott and Bilyk 1996). In the Suffield National Wildlife Area in southern Alberta, long-billed curlews were

most abundant in upland grassland and disturbed (*i.e.*, formerly cultivated, mowed or heavily grazed) grassland as well as moist grasslands (Dale *et al.* 1999).

As part of the Multiple Species at Risk (MULTISAR) conservation program in southern Alberta, Habitat Suitability Index (HSI) models were developed for various wildlife species at risk inhabiting the Milk River Watershed. Using information on known locations as well as the biology and ecology of wildlife species, HSI models use various physical and habitat variables to predict where suitable habitat for the species may be found on the landscape. Three habitat variables were included in the HSI developed for the long-billed curlew: percent native prairie (native grass cover) by quarter-section; shrub cover; and topography (Downey 2004a). According to the model, long-billed curlews prefer native prairie with less than 10% shrub cover on flat terrain (*i.e.*, with slopes less than 15°). Also as part of the MULTISAR program, a second HSI model was developed for long-billed curlew habitat use in the headwaters of the Oldman River Watershed (Downey 2004b). This model included the same three variables as the aforementioned model, but also included tree cover, with long-billed curlews preferring tree cover between 0% and 5%.

1.3.2 Nesting Habitat

Long-billed curlew nesting habitat characteristics vary throughout their range; however, in general, curlews nest in open expanses of short (*i.e.*, less than 10 cm to 20 cm) grassland with flat to rolling topography (Allen 1980, Hill 1998, Dechant *et al.* 2002a, Dugger and Dugger 2002). Long-billed curlews tend to avoid nesting in treed areas, grasslands with a high density of shrubs or in tall *(i.e.*, greater than 30 cm), dense grasslands (Pampush and Anthony 1993, Dechant *et al.* 2002a, Dugger and Dugger 2002). Although long-billed curlews will occasionally nest in hayland and cropland (including fallow and stubble fields) with a similar vertical profile as shortgrass prairie, extensive areas of cultivated land are not considered preferred nesting habitats (Renaud 1980, De Smet 1992a, Hill 1998, Dechant *et al.* 2002a).

Four important nesting habitat requirements for long-billed curlews identified in Oregon, included: 1) short grass (*i.e.*, generally less than 30 cm tall); 2) bare ground; 3) shade; and 4) abundant invertebrate prey (Pampush and Anthony 1993). In Oregon, long-billed curlew nest density was negatively correlated with vegetation height and vertical density (Pampush and Anthony 1993). Long-billed curlews preferentially nested in exotic downy brome (*Bromus tectorum*) habitats and avoided native bunchgrass habitats (Pampush and Anthony 1993). Allen (1980) also documented a preference for long-billed curlews to nest in downy brome vegetation less than 10 cm tall in southeastern Washington. In Utah, long-billed curlew nest sites had significantly shorter vegetation than in surrounding areas (5.7 cm versus 9.0 cm) (Paton and Dalton 1994). Low shrub cover and a low vegetative profile are suspected to be important for predator detection and avoidance as well as effective communication between nesting birds (Allen 1980, Pampush and Anthony 1993). Vegetation patchiness at nest sites (*i.e.*, uneven vegetation height and the irregular spacing of grass clumps) and the placement of nests near conspicuous objects such as livestock manure piles, rocks, or dirt mounds may also be important for predator avoidance (Allen 1980, Pampush and Anthony 1993, Dugger and Dugger 2002).

1.3.3 Brood-rearing Habitat

Although long-billed curlews usually nest in areas with shorter vegetation, curlew broods require areas with taller, denser grass, to provide shade and cover from predators (Maher 1973, Allen 1980, Hill 1998, Dugger and Dugger 2002). Adults move broods to areas with higher vegetation shortly after eggs hatch; one brood was reported to have moved more than 6 km in a six-day period (Maher 1973). Chicks usually leave the nest within a few hours of hatching (Dugger and Dugger 2002). Long-billed curlew broods are known to use cropland, stubble fields and weedy areas to a greater extent during the brood rearing period, especially if vegetation at the nest site is sparse (Maher 1973, 1974, Allen 1980, Renaud 1980, Pampush and Anthony 1993, Dechant *et al.* 2002a). In Nevada, long-billed curlews preferred hay fields to rangeland for brood-rearing (Hartman and Oring 2009).

1.3.4 Foraging Habitat

In general, open, sparse grasslands offer preferred foraging habitat for adult long-billed curlews (Dechant *et al.* 2002a, Dugger and Dugger 2002). It is suspected that short vegetation (less than 10 cm) facilitates predator detection and is easier for a long-billed bird to forage in than in tall, dense vegetation (Redmond 1986, Dugger and Dugger 2002). Breeders may forage within or outside nesting territories (Pampush and Anthony 1993). Long-billed curlews are known to forage in grasslands, cultivated fields, stubble fields and wet meadows (Dechant *et al*. 2002a).

In shortgrass prairie and tame pasture in Idaho, Bicak *et al.* (1982) found that long-billed curlew prey capture rates were higher in areas with shorter grass (*i.e.*, less than 10 cm), despite higher prey densities in taller vegetation. Similarly, Redmond (1986) found that long-billed curlews in Idaho foraged mainly in shortgrass prairie with vegetation heights of 3.6 cm to 9.7 cm. A year of high spring rainfall resulting in tall (*i.e.*, 10 cm to 40 cm), dense grass cover and thick, standing-dead vegetation, hindered long-billed curlew foraging, prompting curlews to travel up to 10 km from territories to forage in agricultural lands (Redmond 1986). Resultant increased energy expenditure is thought to have negatively affected long-billed curlew productivity that year (Redmond 1986). In Oregon, Pampush and Anthony (1993) noted that long-billed curlews used swathed alfalfa (*Medicago sativa*) as a foraging area, but not as a nesting area. Alfalfa fields were used for foraging until plant growth exceeded 30 cm (Pampush and Anthony 1993).

Long-billed curlews may forage singly or in groups of three to 14 (Dugger and Dugger 2002). If there is a high density of grasshoppers, they are most likely to forage in groups (Dugger and Dugger 2002).

1.3.5 Area Requirements

Area requirements for long-billed curlews appear to be associated with the type and amount of vegetation and topography (Dechant *et al*. 2002a). A breeding long-billed curlew pair will have a territory size of approximately 6 ha to 20 ha during the nesting season (Dechant *et al.* 2002a). In southeastern Washington, areas with open, flat, less diverse habitat supported larger long-billed curlew territories than did areas with more diverse topography and that had shrubby vegetation near the nest sites; territories ranged from 14 ha to 20 ha in the former habitat and from 6 ha to 8 ha in the latter habitat (Allen 1980). Dechant *et al.* (2002a) suggest that long-billed curlews

require a minimum habitat area of at least 42 ha, which is approximately three times as large as the average size of a long-billed curlew territory (*i.e.*, approximately 14 ha). Hartman and Oring (2009) observed mean nest densities of 4.3, 4.7 and 1.3 nests/100 ha in Nevada hay fields, open rangeland and shrub-desert rangeland habitat types, respectively. Saunders (2001) reported a mean long-billed curlew density of 0.18 pairs/100 ha in quarter-sections with 51% to 100% native grassland in southern Alberta. Long-billed curlew densities were approximately two times lower in quarter-sections with less than 50% native prairie. In East Kootenay, British Columbia, long-billed curlews only nested in grassland openings that were greater than 250 m in diameter (Ohanjanian 1992, cited in COSEWIC 2002a).

2 UPLAND SANDPIPER

2.1 Background

The upland sandpiper is a relatively indistinct shorebird with a slender neck, rounded head, dark-coloured 'Vs' on its breast and a light median stripe on its crown (Semenchuk 1992). Similar to long-billed curlews, upland sandpipers are shorebirds that are specialized to breed in dry upland grasslands (Houston and Bowen 2001). The upland sandpiper is a regular breeder in the Grassland Natural Region of Alberta (Semenchuk 1992, NRC 2006). This species of sandpiper is completely terrestrial and is not associated with wetlands or coastal areas (Houston *et al.* 2011). The main, contiguous portion of the upland sandpiper breeding range extends south from southern Canada to the central United States and east from the Rocky Mountains to the Appalachian Mountain region (Houston and Bowen 2001). The majority (*i.e.*, approximately 79%) of the continental population of upland sandpipers is found in South Dakota, North Dakota, Nebraska and Kansas. Small, isolated breeding populations occur in high-altitude meadows in Washington, Oregon, Idaho, Alaska and Yukon as well as the southwest corner of the Northwest Territories. Within Canada, upland sandpipers breed in the southern regions of Alberta, Saskatchewan and Manitoba. Upland sandpipers winter in South America.

Upland sandpipers are currently listed as 'Sensitive' in Alberta under the General Status listing (GoA 2016). The species is not listed provincially under the *Wildlife Act* except as a 'non-game animal'. The upland sandpiper has not been assessed nationally by COSEWIC and it is not listed under Schedule 1 of *SARA* (COSEWIC 2016). Internationally, the upland sandpiper is listed as a species of 'Least Concern' by the IUCN (BirdLife International 2012b).

Initially abundant across the grasslands of North America, upland sandpiper populations plummeted during the late 1880's and early 1900's throughout their former range (Houston 1999). Breeding Bird Surveys from 1966 to 1999 suggest that upland sandpiper populations are continuing to decline throughout North America, with the possible exception of North Dakota (Houston and Bowen 2001). In 2009, the estimated Canadian population of upland sandpipers was approximately 43,000 individuals (Prairie Habitat Joint Venture 2009).

Overhunting in its breeding and wintering grounds in the 1800's and early 1900's and conversion of vast tracts of native prairie habitat to cropland are thought to have played a major role in the initial decline of this species (Houston 1999, Houston and Bowen 2001). Habitat loss,

fragmentation and degradation due to human activities continue to have a negative impact on upland sandpiper populations throughout much of their breeding range, including in Alberta (Buhnerkempe and Westemeier 1988, Helzer and Jelinski 1999, Houston and Bowen 2001, GoA 2016).

2.2 Ecology

Upland sandpipers arrive in Alberta from early to mid-May, and depart by late August or early September (Semenchuk 1992). Breeding pairs usually arrive on breeding grounds approximately 14 days prior to nest initiation (Houston and Bowen 2001). Pairs typically arrive together, or form shortly after arrival (Houston and Bowen 2001). Upland sandpipers often return to the same breeding site in subsequent years (Ailes 1976, 1980, Dorio 1977, Houston and Bowen 2001). In Alberta, nesting usually occurs from mid-May to the end of June, and brood-rearing takes place from June until early August (Kantrud and Higgins 1992, Semenchuk 1992, Houston and Bowen 2001).

Upland sandpipers are typically non-territorial, seldom nesting alone and more often nesting in loose colonies (Houston and Bowen 2001). Feeding and loafing areas are usually shared with other adults and broods (Ailes 1980, Houston and Bowen 2001). Normally, four eggs are laid, with clutch sizes ranging from two to seven eggs (Houston and Bowen 2001). Most commonly, one brood is raised per season (Semenchuk 1992). Eggs are usually incubated for a period of 23 to 24 days (Houston and Bowen 2001). Both sexes share incubation duties, although the male is often more persistent (Ailes 1976). Chicks are precocial and leave the nest shortly after hatching, after which they are tended by both parents (Semenchuk 1992). Young fledge within 32 to 34 days (Ailes 1980).

2.2.1 Diet

Upland sandpipers feed mainly on terrestrial insects and other invertebrates, including grasshoppers, weevils, spiders, snails and earthworms (Houston and Bowen 2001). Lesser amounts of waste grains and weed seeds are also eaten (Houston and Bowen 2001).

2.2.2 Predators

Upland sandpiper adults, eggs and chicks are most commonly susceptible to mammalian predators such as coyotes, American badgers, common raccoons (*Procyon lotor*), striped skunks (*Mephitis mephitis*), minks (*Mustela vison*) and red foxes (Houston and Bowen 2001). Common avian predators include: American crows, golden eagles (*Aquila chrysaetos*), northern goshawks (*Accipiter gentilis*), sharp-shinned hawks (*A. striatus*), Cooper's hawks (*A. cooperii*), northern harriers (*Circus cyaneus*), American kestrels (*Falco sparverius*) and snowy owls (*Nyctea scandiaca*) (Houston and Bowen 2001).

Out of 617 nests in South Dakota, North Dakota, Montana and Manitoba, 32% (197 nests) were destroyed, with mammalian predators responsible for 66% of losses (Kantrud and Higgins 1992). Avian predators were responsible for significantly fewer (3%) nest losses (Kantrud and Higgins 1992).

2.3 Habitat Requirements

2.3.1 General

In general, upland sandpipers use habitats with "low to moderate forb cover, low woody cover [*i.e.*, minimal shrub or tree growth], moderate grass cover, moderate to high litter cover, and little bare ground" (Dechant *et al.* 2002b). Fence posts and other display perches are also considered important habitat components (Dechant *et al.* 2002b). Upland sandpipers utilize native prairie, tame pasture, hay fields and wet meadows, as well as cropland and grassy rights-of-way adjacent to roads or railroads (Dechant *et al.* 2002b). Native prairie habitats are typically preferred over cultivated lands (Kantrud and Higgins 1992). Native prairie appears to provide a more structurally-suitable, secure and potentially more productive habitat than cultivated fields (Kirsh and Higgins 1976, Ailes 1980, Kantrud and Higgins 1992, Dechant *et al.* 2002b). Upland sandpipers in Manitoba were found to be significantly more abundant in native mixedgrass prairie than in sites dominated by non-native species, such as smooth brome (*Bromus inermis* ssp. *inermis*), Kentucky bluegrass (*Poa pratensis*) and leafy spurge (*Euphorbia esula*). In Saskatchewan, upland sandpipers were present in low abundance in both native mixedgrass and crested wheat grass (*Agropyron cristatum*) tame pastures (Sutter and Brigham 1998). In the Hand Hills of southern Alberta, upland sandpipers were 2.3 times more abundant in native fescue (*Festuca hallii*) grassland than in cultivated areas (Owens and Myres 1973).

Upland sandpipers use habitats with a range of vegetation heights over the course of the breeding season (Dechant *et al.* 2002b). Areas with short vegetation (*i.e.*, less than 30 cm) are used for foraging (Dorio 1977, Dorio and Grewe 1979, Bolster 1990); taller vegetation (*i.e.*, 10 cm to 64 cm) provides suitable cover for nesting (Higgins *et al.* 1969, Ailes 1976, Kaiser 1979, Buhnerkempe and Westemeier 1988); and short to intermediate vegetation (*i.e.*, less than 15 cm) is used for brood-rearing (Ailes 1976, Dorio 1977, Dorio and Grewe 1979, Ailes 1980, Buhnerkempe and Westemeier 1988, Bolster 1990). To meet these habitat requirements, upland sandpipers will use areas with varying amounts and types of disturbance; grazed, burned and hayed fields generally provide suitable foraging and brood-rearing habitats, while undisturbed or lightly grazed areas are used for nesting (Dechant *et al.* 2002b).

2.3.2 Nesting Habitat

Upland sandpipers are ground nesters that have been found to nest in a variety of habitats, including native and non-native vegetation, with varying vegetation heights and densities (Houston and Bowen 2001, Dechant *et al.* 2002b). Vegetation structure rather than plant species composition is thought to be important in nest site selection (Dorio and Grewe 1979, Patterson and Best 1996, Dechant *et al.* 2002b). Several studies have examined nesting habitat preferences of upland sandpipers within its breeding range in the United States (Dechant *et al.* 2002b). Comparatively few studies have been conducted in Canada.

Kantrud and Higgins (1992) summarized the characteristics of upland sandpiper nests found in North and South Dakota, Montana and southern Manitoba during 1963 to 1991. Most nests (84%, or 366 of 434) were located in native prairie, while 13% (58 of 434) of nests were found in seeded grasslands (including tame pastures, haylands and seeded native grasses) and less than 1% (10 of 434) of nests were found in cropland (including stubble and fallow fields but not no-

till or minimum tillage) (Kantrud and Higgins 1992). In terms of land-use, 49% (219 of 445) of nests were located in grazed pastures, 40% (179 of 445) of nests were located in idle or ungrazed grasslands, 8% (37 of 445) of nests were situated in crop fields and less than 1% (10 of 445) of nests were found in hayfields (Kantrud and Higgins 1992). Nests were most often placed in grass-dominated sites with intermediate cover height (mean 26 cm) and density. The majority of nest sites had 100% visual obstruction at less than 15 cm, and effective cover heights of less than 30 cm. In south-central North Dakota, upland sandpiper nests were most commonly found in areas with greater than 50% grass cover and less than 50% forb cover (Bowen and Kruse 1993). In Illinois, upland sandpipers most frequently nested in cover from 17 cm to 33 cm tall, in weedy fields that had been rotary mowed the previous summer or in old redtop – timothy (*Agrostis alba – Phleum pretense*) meadows invaded with forbs and Kentucky bluegrass (Buhnerkempe and Westemeier 1988). Upland sandpipers tended to avoid nesting in fields with relatively uniform grass or legume cover (Buhnerkempe and Westemeier 1988). In central Wisconsin, Ailes (1980) found that upland sandpipers avoided nesting in heavily grazed grasslands or small-grain agriculture; instead, most nests were observed in hayfields or idle fields, with 54% of nests located in vegetation cover 25 cm to 40 cm tall (Ailes 1980). Grasslands in this study were dominated by Kentucky bluegrass, quack grass (*Elymus repens*), timothy and smooth brome. Of 15 nests in central Minnesota, the majority were in old crop fields (73%), whereas 20% were situated in tame pastures dominated by smooth brome and 7% were in a sedge – grass (*Carex* spp. – grass) meadow (Dorio and Grewe 1979). Most nests were in vegetation 22.5 cm to 35.0 cm tall (Dorio 1977). In Saskatchewan, upland sandpipers most commonly nested in areas with tall, dense vegetation greater than 15 cm tall (Colwell and Oring 1990).

Forb cover may be an important characteristic of nesting habitat as several studies have documented the tendency of upland sandpipers to nest in areas with moderate amounts of forbs (Buhnerkempe and Westemeier 1988, Dechant *et al.* 2002b). The proximity of nest sites to suitable loafing and foraging habitats and the availability of display perches, such as fence posts, rock piles or tree stumps, are also important habitat-related factors (Buhnerkempe and Westemeier 1988, Dechant *et al.* 2002b).

2.3.3 Brood-rearing Habitat

As with nesting, a range of habitats are used during the brood-rearing period (Dechant *et al.* 2002b). Bolster (1990) noted a movement of broods from pastures to alfalfa fields; alfalfa and small-grain fields with vegetation heights less than 27 cm tall were used more often than expected during brood-rearing. In Illinois, Buhnerkempe and Westemeier (1988) observed broods in wheat stubble, recently hayed legumes, old redtop and moderately grazed pastures. Broods used open, weedy habitats with short (*i.e.*, less than 20 cm tall) vegetation (Buhnerkempe and Westemeier 1988). Openness and low vertical cover are thought to facilitate movement and foraging by chicks (Buhnerkempe and Westemeier 1988). In Wisconsin, Ailes (1980) reported that most upland sandpiper adults and broods moved to heavily grazed pastures during the brood-rearing period (*i.e.*, late June and July). Of the grazed fields used for foraging during brood-rearing, 65% of fields were heavily grazed with vegetation heights of less than 10 cm (Ailes 1976, 1980).

2.3.4 Foraging Habitat

In general, upland sandpipers prefer areas with short vegetation for foraging, potentially due to improved visibility and increased foraging efficiency (Dorio and Grewe 1979, Houston and Bowen 2001, Dechant *et al.* 2002b). In central Minnesota, Dorio and Grewe (1979) reported that upland sandpiper young and adults moved from denser nesting cover to forage in areas of blowouts as well as heavily grazed pastures, both of which are habitats with vegetation heights under 10 cm tall. The majority (54%) of feeding observations during incubation and brood-rearing were in heavily grazed pastures (Dorio and Grewe 1979). Sedge – grass meadows were used during May until vegetation heights exceeded 30 cm tall (Dorio and Grewe 1979). Similarly, out of 1,116 feeding observations in central Wisconsin, the majority (*i.e.*, 66%) were in grazed pastures, 13% were in ungrazed pastures, 11% in hayfields, 6% in cropland and 3% in plowed fields (Ailes 1980). The majority (*i.e.*, 68%) of grazed pastures used for foraging were heavily grazed with vegetation heights of 0 cm to 10 cm.

2.3.5 Area Requirements

Upland sandpipers are considered highly sensitive to habitat fragmentation, with their abundance positively correlated with field or patch size (Vickery *et al.* 1994, Helzer and Jelinski 1999, Walk and Warner 1999). Minimum area requirements of 50 ha to 61 ha (Helzer and Jelinski 1999), 65 ha (Walk and Warner 1999) and 200 ha (Vickery *et al.* 1994) have been reported for upland sandpipers in Nebraska, Illinois and Maine, respectively. It is recommended that habitat patches at least 100 ha in size (preferably 200 ha) and less than 1 km apart are retained and appropriately managed to provide sufficient habitat heterogeneity for upland sandpipers and other grassland species and to reduce edge effects (Herkert 1994, Vickery *et al.* 1994, Walk and Warner 1999). Habitat patches with high amounts of interior area are preferable as studies have shown that nest predation rates are higher in edge habitats (Johnson and Temple 1990).

3 SPRAGUE'S PIPIT

3.1 Background

The Sprague's pipit is a solitary, secretive bird with large, dark eyes, flesh-coloured legs and upper parts that are buff-coloured and distinctly streaked (Semenchuk 1992). The Sprague's pipit reaches its highest continental abundance in southeastern Alberta (Prescott 1997). Although rarely seen, Sprague's pipits are best detected by the males' distinctive high-pitched, descending flight song. They are considered an indicator of grassland health in the Canadian Prairies and therefore thought of as a flagship species for other rare and endangered grassland species (Environment Canada 2012). Sprague's pipits breed primarily in native grasslands of the Canadian prairies and the Northern Great Plains (Prescott 1997, Robbins and Dale 1999). This species is at the northwest corner of its breeding range in Alberta where it breeds most commonly in the Grassland Natural Region and sporadically in the Central Parkland Natural Region (Semenchuk 1992, NRC 2006). Within Canada its breeding range extends east from the foothills of the Rocky Mountains across southern and central Alberta through central and southern Saskatchewan to west-central and southern Manitoba (Robbins and Dale 1999). A single confirmed breeding record has been documented in south-central British Columbia

(McConnell *et al.* 1993). In the United States its breeding range extends east from Montana through North Dakota, north and central South Dakota and into northwestern Minnesota (Prescott1997, Robbins and Dale 1999). Sprague's pipits winter in the southern United States and northern Mexico (Prescott 1997, Robbins and Dale 1999).

Sprague's pipits are currently listed as 'Sensitive' under the Alberta General Status listing (GoA 2016). The species has been designated as a species of 'Special Concern' under the *Wildlife Act*. Nationally, the Sprague's pipit was examined by COSEWIC in 1999 and given a status of 'Threatened' (Prescott and Davis 1999). That status was re-confirmed in 2000 and 2010, and the species remains 'Threatened' to this date (COSEWIC 2000, 2010, 2016). Sprague's pipits are listed as 'Threatened' under Schedule 1 of the *Species at Risk Act*. A Recovery Strategy for the Sprague's pipit was released by the federal government in 2012 (Environment Canada 2012). Internationally, the Sprague's pipit is listed as 'Vulnerable' by the IUCN (BirdLife International 2012c).

Similar to long-billed curlews and upland sandpipers, Sprague's pipits have experienced population declines or extirpations throughout its breeding range (Robbins and Dale 1999). Rapid population declines were reported over much of its breeding range in the decades leading up to the end of the last century (Prescott 1997). North American Breeding Bird Survey trends indicate Canadian and continental population declines of 7% and 5% per year, respectively, between 1966 and 2002 (Sauer *et al.* 2003). Declines of approximately 8% per year were recorded in Alberta over the same period (Sauer *et al.* 2003). The Canadian population for the Sprague's pipit is currently estimated at 720,000 birds (COSEWIC 2010).

The continuing loss, fragmentation and degradation of native prairie habitats are considered the most significant limiting factors to Sprague's pipit populations throughout most of its range (Prescott 1997, Prescott and Davis 1999). Other factors include climate change, predation from mammalian and avian predators and severe weather events (COSEWIC 2010). Conversion of native prairie to cropland, intensive grazing and encroachment of woody vegetation due to reduced fire frequency are factors that can degrade or eliminate suitable Sprague's pipit habitat (Prescott and Davis 1999).

3.2 Ecology

Sprague's pipits arrive on their breeding grounds in Alberta in late April to mid-May (Prescott 1997). Males have a distinctive flight display during the breeding period which often lasts over thirty minutes (Robbins and Dale 1999). Sprague's pipits have defined breeding territories that are used for both nesting and feeding (Robbins and Dale 1999). Nest building usually begins in early to mid-May (Robbins and Dale 1999). Egg-laying initiation dates range from the second week of May to the third week of July (Maher 1973, Robbins and Dale 1999). A clutch size of four or five eggs is typical (Prescott 1997). Ludlow *et al.* (2014) observed a mean clutch size of 4.7 ± 0.6 eggs (range: 4-6) in southeastern Alberta. Females are primarily responsible for incubation and tending to the chicks (Prescott 1997). Eggs are usually incubated from 13 to 14 days (Robbins and Dale 1999). Newly hatched chicks (nestlings) are altricial, meaning their eyes are closed and they have little or no down and are reliant on being fed by their parents (Robbins and Dale 1999). Young leave the nest nine to 12 days after hatching (Maher 1973), with fledging

dates ranging from mid-June to late August in Saskatchewan (Robbins and Dale 1999). Sprague's pipits may raise two broods in a season, although the frequency of re-nesting is thought to be fairly low (Sutter *et al.* 1996, Robbins and Dale 1999). Pipher (2011) observed a nesting success rate (*i.e.*, where nests fledged at least one young) of 46% for Sprague's pipits in Grasslands National Park, and Ludlow *et al.* (2014) observed a nesting success rate of 52% on the Antelope Creek Ranch in southeastern Alberta (Ludlow *et al.* 2014).

In Saskatchewan, flocks of adult and immature Sprague's pipits begin to form in mid-July, with flock sizes increasing toward the end of August and beginning of September prior to migration (Robbins and Dale 1999). Most birds depart for their wintering grounds in mid-September, although Sprague's pipits have been recorded in Saskatchewan as late as the first week of October (Prescott 1997).

3.2.1 Diet

Sprague's pipits are almost entirely insectivorous (Maher 1979, Robbins and Dale 1999). The type of insects consumed varies over the breeding season. Maher (1974) examined food habitats of Sprague's pipits in the mixedgrass prairie of southwestern Saskatchewan during four summers. The author found that beetles comprised more than 40% of the adult diet in May, whereas grasshoppers increased from 4% of the diet in May to 91% in September (Maher 1974). Seeds comprised less than 3% of the adult diet (Maher 1974). Nestlings also consumed a significant proportion of grasshoppers as well as lesser amounts of lepidopteran larvae (butterflies and moths), homopterans (leaf hoppers), aranerans (spiders) and hymenopterans (bees, ants and wasps) (Maher 1974, 1979).

3.2.2 Predators

Predation can have a significant impact on Sprague's pipit nesting success (Prescott 1997). In Saskatchewan, predation was responsible for up to 69% of all Sprague's pipit nest losses (Maher 1973). In this study, 53% and 42% of nests survived the incubation and nestling periods, respectively (Maher 1973). In Grasslands National Park, 46% (*i.e.*, 29 of 63) of Sprague's pipit nests were depredated (Pipher 2011).

Mammalian predators such as mice (*Peromyscus* spp.), ground squirrels (*Spermophilus* spp.), weasels, American badgers, coyotes, red foxes, raccoons and skunks account for a significant proportion of nest losses among grassland-nesting birds (Patterson and Best 1996, Sutter 1997). Avian predators that target adults and young birds include various species of raptors, as described for long-billed curlews and upland sandpipers. Sprague's pipit nest productivity may be reduced by brown-headed cowbirds (*Molothrus ater*), which are known to parasitize (*i.e.*, lay their eggs in) pipit nests (Robbins and Dale 1999, Davis and Sealy 2000). However, Ludlow *et al.* (2014) reported no incidences (n=22) of Sprague's pipit nest parasitism by cowbirds in their study in southeastern Alberta.

3.3 Habitat Requirements

3.3.1 General

Sprague's pipits are primarily found in well-drained, open, fescue, dry mixedgrass and mixedgrass native prairie habitats (Prescott 1997, Robbins and Dale 1999). Numerous studies have found that Sprague's pipits tend to prefer native prairie habitats and show an aversion to tame pasture, hayfields, croplands and wetlands (Owens and Myres 1973, Wilson and Belcher 1989, Dale 1992, Prescott and Bilyk 1996, Prescott and Murphy 1996, Madden 1996, Davis and Duncan 1999, Davis *et al.* 1999, Madden *et al.* 2000, Koper *et al.* 2009). This trend has been confirmed in fescue prairie in the Hand Hills of southern Alberta where Sprague's pipits were significantly more abundant in native prairie than in cultivated lands (Owens and Myres 1973). In a survey of the Southern Prairie Biome of Alberta, Prescott and Bilyk (1996) observed Sprague's pipits in 63% (n=27) of native mixedgrass prairie sites and 44.4% (n=18) of fescue prairie sites. Sprague's pipits were only found in one tame pasture site (n=16) and were not present in cropland or hayfields (Prescott and Bilyk 1996). Similarly, Prescott and Wagner (1996) found that Sprague's pipits were significantly more common in native fescue or mixedgrass prairie than in tame pasture near Brooks, Alberta. In southern Saskatchewan, Davis *et al.* (1999) found that Sprague's pipits were more common in fescue and mixedgrass prairie than in hayland or cropland. Sprague's pipits are also usually absent or less frequent in Permanent Cover Program (PCP) sites in the Canadian prairies. During an evaluation of Agriculture Canada's PCP, McMaster and Davis (2001) reported that Sprague's pipits were infrequently detected in PCP sites in Alberta, Manitoba and Saskatchewan. PCP sites are old crop fields that were seeded to perennial grasses to provide permanent cover. PCP sites were comprised of a combination of wheatgrass (*Agropyron* spp.), brome (*Bromus* spp.), alfalfa and/or crested wheat grass. Koper *et al.* (2009) observed that Sprague's pipits in southern Alberta were observed in higher abundances at greater distances from croplands, forage crops (*i.e.*, tame pastures and hay fields) and wetlands.

In general, vegetation structure is considered more important than plant species composition in determining Sprague's pipit habitat associations (Dechant *et al.* 2002c). Despite their preference for native prairie, Sprague's pipits have not been associated with specific plant species. Sprague's pipit preference for native prairie is thought to be due to the inherent structural heterogeneity of these habitats (Prescott 1997). Sprague's pipits tend to avoid tall vegetation and deep litter and usually occur in areas with intermediate vegetation height and density and moderate to high grass to forb ratios (Prescott and Murphy 1996, Prescott 1997, Madden *et al.* 2000, Dechant *et al.* 2002c). Sprague's pipits have been positively associated with moderate to high grass cover (33% to 53%), maximum vegetation heights of 28 cm and litter depths of 1.2 cm to 3.1 cm (Dale 1983, Hooper and Pitt 1996, Madden 1996, Sutter 1996, Schneider 1998, Madden *et al.* 2000). In contrast to these findings, in the Milk River Watershed of southwestern Saskatchewan, Sprague's pipit probability of occurrence increased with higher vegetation volume and litter mass (Henderson and Davis 2014). The reason for this apparent disagreement is not clear. Sprague's pipits also typically occur in areas with low shrub cover (Dechant *et al.* 2002c). Studies in North Dakota and southern Saskatchewan have found a negative association between Sprague's pipits and woody vegetation (Faanes 1983), shrubs 20 cm to 100 cm tall (Anstey *et al.* 1995) and density of low growing shrubs (Schneider 1998), as well as a positive association with low visual obstruction (Madden 1996, Madden *et al.* 2000). Madden *et al.*

(2000) reported that Sprague's pipits in mixedgrass prairie in North Dakota were most common at visual obstruction readings of less than 8 cm.

Similar to the long-billed curlew, an HSI model was developed by MULTISAR for Sprague's pipits in the Milk River Watershed (Landry 2004a). This HSI model included three, equally-weighted variables: percent native prairie (grass cover) by quarter-section; percent tree and shrub cover; and distance from riparian areas. According to this model, areas comprised of greater than 25% native grassland, less than 10% shrub cover and located away from riparian areas were rated as highly suitable Sprague's pipit habitat. The HSI model was limited by the difficulty of modeling micro-habitat requirements such as grass height and density. A second HSI model was developed Sprague`s pipit habitat use in the headwaters of the Oldman River Watershed (Landry 2004b). The same three variables as the original model were used with similar results.

3.3.2 Nesting Habitat

Sprague's pipits construct nests on the ground, usually at the base of dense clumps of grass in open native prairie with vegetation heights between 10 cm to 30 cm (Robbins and Dale 1999). In southwestern Saskatchewan, Sutter (1997) found that Sprague's pipits preferred to nest in sites with dense, grassy and relatively tall vegetation with low forb density and little bare ground in comparison to random sites. These characteristics are thought to offer protection from predation as well as reduced heat stress (Sutter 1997). Average, maximum vegetation height at 47 nests was approximately 28 cm, with litter depth averaging 2.4 cm (Sutter 1997). Most nests were either completely or partially domed with a grass canopy comprised most often of northern wheat grass (*Elymus lanceolatus*), the dominant grass at the study site. Also in southern Saskatchewan, in Grasslands National Park Sprague's pipits chose nest sites where vegetation was significantly taller and denser, litter depth was greater and there was less biocrust and bare ground cover than at random locations within 50 m (Pipher 2011). In the same study area, Lusk (2011) found that Sprague's pipit daily nest survival declined with increased vegetation density and litter depth at the nest site. In Montana, Sprague's pipit nest sites had relatively tall, dense vegetation with high, thick cover of litter (Dieni and Jones 2003).

3.3.3 Foraging Habitat

Throughout the year, Sprague's pipits forage on the ground in open grassland habitats (Robbins and Dale 1999). Foraging occurs within the breeding territory.

3.3.4 Area Requirements

Area sensitivity of Sprague's pipits is a topic that requires additional research in the Milk River and South Saskatchewan River watersheds. Prescott and Davis (1999) suggest that areas of suitable habitat should be greater than 150 ha in size in order to be attractive as breeding sites for Sprague's pipits. In Saskatchewan, Sprague's pipits are area sensitive and require a minimum area of 145 ha to 190 ha (Saskatchewan Wetland Conservation Corporation [SWCC] 1997, Davis 2004). Patch size may have an impact on rates of brown-headed cowbird parasitism. In Manitoba, brown-headed cowbird brood parasitism was higher on smaller tracts of land (22 ha) in comparison to larger habitat patches (64 ha) (Davis and Sealy 2000).

4 BAIRD'S SPARROW

4.1 Background

The Baird's sparrow is a shy, wary sparrow with a short tail, buff-coloured head and breast and black streaking on the breast (Semenchuk 1992). Its breeding range, like that of the Sprague's pipit, is confined to the southern grassland regions of the Canadian Prairies and the adjoining states to the south. Within Alberta, Baird's sparrows breed most commonly south of Stettler and east of the Red Deer River, with an extension west to the Calgary Area (Semenchuk 1992). The main Baird's sparrow breeding range extends from the southern regions of Alberta and Saskatchewan and the southwest portion of Manitoba, south to central and eastern Montana, and east to North Dakota and north-western and north-central South Dakota (Green *et al.* 2002). Baird's sparrows winter in the extreme southern United States (Arizona, Texas and New Mexico) into north-central Mexico (Green *et al.* 2002).

The Baird's sparrow is currently listed as 'Sensitive' under the General Status listing for Alberta wildlife (GoA 2016). The species has not been designated as a species at risk under the provincial *Wildlife Act*; however, it is listed as a 'non-game animal'. Nationally, the Baird's sparrow was originally listed as 'Threatened' in 1989, de-listed to 'Not at Risk' in 1986, and up-listed to 'Special Concern' status in 2012, where it remains today (COSEWIC 1996, 2012, 2016). Internationally, the Baird's sparrow has been designated as a species of 'Least Concern' by the IUCN (BirdLife International 2012d).

As with Sprague's pipits, there is limited population and trend information available for Baird's sparrows in Alberta and an accurate estimate of population size is unavailable for this province (Green *et al.* 2002). In 1994, the estimated population of singing males in Saskatchewan was 960,000 (Davis *et al.* 1996). The current global population of this species is estimated to be 1.2 million birds (± 50%), of which 60% breed in Canada (COSEWIC 2012). Baird's sparrows have experienced a 25% population decline in Canada in the last decade (COSEWIC 2012). The species has experienced large population declines in Alberta since the mid-1990s (GoA 2016). Contraction of its breeding range has also been reported in both Minnesota and southeastern Manitoba (Green *et al.* 2002).

As with the previous species discussed, Baird's sparrow populations are vulnerable to habitat loss, degradation and fragmentation (Davis and Sealy 1998, Green *et al.* 2002). Factors that contribute to habitat degradation and fragmentation and make habitat unsuitable for this grassland-dependent species include: conversion of native prairie to cropland and non-native vegetation; invasion of native prairie by exotic plants; shrub encroachment due to fire suppression; and unsustainable range management practices leading to over-grazing (Owens and Myres 1973, Goosen *et al.* 1993, Green *et al.* 2002, GoA 2016).

4.2 Ecology

Baird's sparrows are considered partially nomadic, due to significant spatial shifts in population densities from one year to the next (Green *et al.* 2002). This behaviour is thought to be an evolved response to the fluctuating spatial and temporal habitat conditions that would have been created by historic fire, drought and bison movements and grazing patterns (Green *et al.* 2002).

Baird's sparrows usually arrive on their breeding grounds in Alberta in mid-May (Semenchuk 1992**,** Green *et al.* 2002). Females often arrive up to one week after males, with pair bonds forming after breeding territories have been established (Green *et al.* 2002). Nest building usually begins by the third week of May, with egg-laying initiated by late May to early June (Davis and Sealy 1998, Green *et al.* 2002). Usually, four to five eggs are laid (Wiggins 2006). Ludlow *et al.* (2014) observed a mean clutch size of 4.4 ± 0.8 eggs (range: 3-6) in southeastern Alberta. Females incubate the eggs for a period of 11 to 12 days (Green *et al.* 2002). Normally, one brood is reared per season; however, confirmed double broods have been documented in southwestern Manitoba (Davis and Sealy 1998). Second clutches were initiated in mid- to late July in Manitoba (Davis and Sealy 1998). Like Sprague's pipit young, Baird's sparrow chicks are altricial, relying on their parents for food. Young fledge and leave the nest eight to 11 days after hatching and are able to take short flights by 13 days old; newly fledged young remain in the parent's territory until 19 days old (Davis and Sealy 1998, Green *et al.* 2002). Pipher (2011) observed a nesting success rate of 75% in Grasslands National Park in Saskatchewan, meaning that three-quarters of all nests produced at least one young. Ludlow *et al.* (2014) observed a nesting success rate of 31% on the Antelope Creek Ranch in southeastern Alberta. Baird's sparrows depart for wintering grounds from mid-September to October (Maher 1973, Dechant *et al.* 2002d).

4.2.1 Diet

During the breeding season, the Baird's sparrow diet consists mainly of insects, including beetles (Coleoptera), grasshoppers (Orthoptera) and caterpillars (Lepidoptera larvae) (Green *et al.* 2002). Various grass and weed seeds and waste grains are also consumed (Green *et al.* 2002). Maher (1979) found that Baird's sparrow nestling diets in southwestern Saskatchewan were comprised mostly of grasshoppers and spiders.

4.2.2 Predators

As with Sprague's pipits, nest predation is the primary cause of nest failure for Baird's sparrows (Green *et al.* 2002). Predators destroyed 39% of 74 nests in southwestern Manitoba (Davis and Sealy 1998), 63% of 167 nests in southern Saskatchewan and 37% of 52 nests in north-central Montana (Green *et al.* 2002). However, in Grasslands National Park, Pipher (2011) observed low predation rates of Baird's sparrow nests (*i.e.*, 19%). Baird's sparrow eggs and young are susceptible to a similar suite of predators as Sprague's pipits (see Section 3.2.2). Nest losses in southwestern Alberta and Manitoba have been attributed to striped skunks, thirteen-lined ground squirrels (*Spermophilus tridecemlineatus*) and Richardson's ground squirrels (*S. richardsonii*) (Mahon 1995, Davis and Sealy 1998). Northern harriers and merlins (*Falco columbianus*) are known predators of young birds (Green *et al.* 2002).

Cowbird brood parasitism is another significant cause of reduced Baird's sparrow productivity (Green *et al.* 2002). Cowbirds parasitized 36% of 74 nests in Manitoba (Davis and Sealy 1998), 21% of 182 nests in Saskatchewan (Green *et al.* 2002) and 6% of 35 nests in southeastern Alberta (Ludlow *et al.* 2014).

4.3 Habitat Requirements

4.3.1 General

Baird's sparrows breed in mixedgrass and fescue prairies with scattered low shrubs (Green *et al.* 2002). Although previously thought to be exclusively associated with native prairie habitats, studies have found that Baird's sparrows do occur in modified grasslands, tame pastures and hayfields with habitat structure components resembling native prairie (Sutter and Brigham 1998, Davis *et al.* 1999, Green *et al.* 2002, Dechant *et al.* 2002d). Baird's sparrows tend to avoid or occur in low densities in cropland (De Smet and Conrad 1991, Davis *et al.* 1996, Davis *et al.* 1999, McMaster and Davis 2001). Actively cultivated lands are considered unsuitable, unproductive habitats (Green *et al.* 2002).

In the Hand Hills of southern Alberta, Owens and Myres (1973) found that Baird's sparrows were significantly more common in native fescue grasslands than in cultivated lands. In the Southern Prairie Biome of Alberta, Baird's sparrows were most common in fescue prairie and to a lesser extent in native mixedgrass prairie, tame pastures and silver sagebrush (*Artemisia cana*) flats (Prescott and Bilyk 1996). Baird's sparrows were only observed in 8.3% of hayfield sites and were absent from cropland (Prescott and Bilyk 1996). In southwestern Alberta, Baird's sparrows in the Milk River Ridge area were found to breed in sites dominated by either Foothills rough fescue (*Festuca campestris*) or crested wheat grass with similar vegetation structure (Mahon 1995). In Saskatchewan, Baird's sparrows were most common in the moist mixed grassland ecoregion and occurred as frequently in hayland as in native and tame pastures, but were significantly less common in cropland (Davis *et al.* 1999). Similarly, Davis *et al.* (1996), Sutter and Brigham (1998) and Davis and Duncan (1999) reported that Baird's sparrows in southern Saskatchewan were as frequent in crested wheat grass pastures as in native prairie habitats. Although hayland or tame pasture dominated by narrow-leaved (2 mm to 4 mm) graminoids such as crested wheat grass are readily used by Baird's sparrows, various studies have found that this species avoid stands of broad-leaved (5 mm to 10 mm) graminoids such as smooth brome (Dale 1992, Anstey *et al.* 1995, Mahon 1995, Madden 1996).

Like Sprague's pipits, Baird's sparrows are thought to be strongly influenced by vegetation structure rather than plant species composition (Mahon 1995, Davis *et al.* 1996, Dechant *et al.* 2002d). Baird's sparrows have been positively associated with grasslands dominated by native or narrow-leaved exotic grasses, shrub cover of less than 20%, litter depths of 0.1 cm to 4 cm and average grass heights of 10 cm to 30 cm (Dale 1983, Sousa and McDonal 1983, Madden *et al.* 2000, Green *et al.* 2002). As with Sprague's pipits, Baird's sparrows tend to avoid grasslands with high amounts of shrubby or woody cover (Green *et al.* 2002, Dechant *et al.* 2002d). In the mixedgrass prairie of northwest North Dakota, Baird's sparrows occurred more commonly in areas with more than 42% grass cover and more than 35% forb cover and with visual obstruction readings of less than 15 cm (Madden *et al.* 2000). In the Milk River Watershed of southwestern Saskatchewan, Baird's sparrows were positively associated with vegetation volume and litter mass and negatively associated with bare ground cover (Henderson and Davis 2014).

4.3.2 Nesting Habitat

Similar to the Sprague's pipit, Baird's sparrows are ground-nesters, often nesting between or beneath clumps of grass. In comparison to random sites, nests in mixedgrass prairie in southern Saskatchewan were found in areas with less bare ground, taller vegetation (average height of 28.5 cm versus 20.1 cm) and increased litter depth (1.1 cm versus 0.5 cm) (Green *et al.* 2002). Nest sites were also characterized by a greater density of standing dead vegetation and lower density of live grasses up to 10 cm high (Green *et al.* 2002). In Grasslands National Park, Baird's sparrows chose nest sites where vegetation was significantly taller and denser, litter depth was greater and there was less biocrust and bare ground cover than at random locations within 50 m (Pipher 2011). In comparison to other prairie songbirds, Baird's sparrows chose nest sites with the tallest, most dense vegetation, highest grass cover and lowest forb cover (Pipher 2011).

In mixedgrass prairie in Montana, Baird's sparrows used nesting habitat with greater litter depth and taller vegetation than expected based on availability (Dieni and Jones 2003). Baird's sparrows also selected nest sites with higher grass and litter cover. The habitat within the immediate vicinity of the nest (*i.e.*, within a 5 m radius) had low bare soil and prairie selaginella cover. Nest sites and surrounding habitat had vertically-denser vegetation than random sites. Western wheat grass (*Pascopyrum smithii*) was preferred over blue grama (*Bouteloua gracilis*) as a nesting substrate.

In the Milk River Ridge area of Alberta, Mahon (1995) observed that Baird's sparrows consistently chose breeding territories with high forb cover and high densities of low and middle canopy grasses (*i.e.*, less than 20 cm) during June and July. Areas with suitable perch sites, such as tall grass clumps or tall forbs and that offered sufficient cover for adults and nests, were preferred (Mahon 1995). In contrast to the findings above, Baird's sparrows in the Milk River Ridge area tended to avoid areas with high shrub cover, high litter depth and vegetation heights exceeding 20 cm (Mahon 1995). The avoidance of tall vegetation and high litter depth in the fescue grasslands of the Milk River Ridge area, in contrast to mixedgrass prairie, may have to do with the inherently higher productivity, tall vegetation and higher litter depths in fescue prairie compared to mixedgrass prairie.

4.3.3 Foraging Habitat

Baird's sparrows forage mostly on the ground in between grass clumps, avoiding open areas (Green *et al.* 2002). Areas with deep litter, dense standing dead vegetation or dense grass cover may impede foraging efficiency by Baird's sparrows (Mahon 1995).

4.3.4 Area Requirements

Multiple studies show that Baird's sparrow territory size ranges between 0.4 ha and 1.5 ha (Wiggins 2006). In Saskatchewan, Baird's sparrow abundance and occurrence was positively associated with patch size of native prairie (Green *et al.* 2002). Baird's sparrows were most common in habitat patches greater than 58 ha (Green *et al.* 2002). Davis (2004) found that Baird's sparrows in Saskatchewan are area sensitive and require a minimum of 25 ha of native mixedgrass prairie habitat. Similarly, Baird's sparrows were positively correlated with the total area of seeded fields in Saskatchewan, and occurred more frequently in PCP lands surrounded by

grasslands versus cropland, wetland, woodland or human residences (McMaster and Davis 2001). Like Sprague's pipits, Baird's sparrows may also be susceptible to higher rates of nest predation and cowbird parasitism in smaller habitat patches (Davis and Sealy 1998). In Saskatchewan, the frequency of cowbird parasitism was greatest among Baird's sparrow nests in native prairie fields that were less than 256 ha (Green *et al.* 2002). High rates of Baird's sparrow brood parasitism were found in Manitoba in native and seeded habitat patches 20 ha to 64 ha in size (Davis and Sealy 1998).

5 GRASSHOPPER SPARROW

5.1 Background

The grasshopper sparrow is a small, inconspicuous sparrow with orange-yellow lores, a darkish crown, a flat forehead and a short tail (Vickery 1996). Males sing a characteristic buzzy, grasshopper-like song from the highest perch in its territory, which is generally a tall shrub, weed stem or fence post (Semenchuk 1992). Grasshopper sparrows occur throughout much of North America, from the southern Prairie Provinces, through the United States to Mexico (Vickery 1996). It is more common in the central and eastern United States than the western part of the country, where it occurs sporadically (Vickery 1996). In Alberta, the species occurs almost exclusively in the Grassland Natural Region with limited occurrences in the Parkland and Boreal Forest Natural Regions (Semenchuk 1992, Federation of Alberta Naturalists 2007).

The grasshopper sparrow is currently listed as 'Sensitive' under the Alberta General status listing (GoA 2016). The species is not listed under the provincial *Wildlife Act* except as a 'non-game animal'. The grasshopper sparrow has not been assessed by COSEWIC and the species is not included under Schedule 1 of *SARA* (COSEWIC 2016). Internationally, the grasshopper sparrow has been designated as a species of 'Least Concern' by the IUCN (BirdLife International 2012e).

Although the grasshopper sparrow appears to be widely distributed across much of temperate North America, it is often locally distributed and even uncommon to rare throughout parts of its range (Vickery 1996). Grasshopper sparrows have been declining in North America since 1994 (GoA 2016). The species may be on the decline in Alberta as well, with one population estimate of less than 10,000 birds province-wide (GoA 2016). Grasshopper sparrows are threatened by cultivation of native prairie and associated disturbances (GoA 2016). Formal status reports have yet to be prepared for the grasshopper sparrow by either Alberta Environment and Parks (AEP) or COSEWIC.

5.2 Ecology

Grasshopper sparrows in Alberta typically arrive at their nesting grounds in the last half of May (Semenchuk 1992). In Saskatchewan and Manitoba, grasshopper sparrows arrive on the breeding grounds in mid-May and depart for their wintering grounds in August (Knapton 1979). Breeding occurs from May to August in Alberta (Federation of Alberta Naturalists 2007). Grasshopper sparrows construct dome-shaped nests in small depressions at the bases of clumps of grass (Whitmore 1981, Semenchuk 1992). Nests are constructed of grass and are usually well-hidden from potential predators (Semenchuk 1992). Nests in the Flint Hills region of Kansas and

Oklahoma were generally oriented with the opening toward the northeast (average = 33.4°), which was against the prevailing southerly wind direction (Long *et al.* 2009). Grasshopper sparrow nest success varies depending on geographic location, ranging from less than 25% in Florida to 54% in Kentucky (Delisle and Savidge 1996, Vickery 1996, Sutter and Ritchison 2005). Nest predation has been identified as the primary factor responsible for reproductive failure in grassland birds in general (Best *et al.* 1997, Koford 1999, Jones *et al.* 2010) and grasshopper sparrows in particular (Vickery 1996, Sutter and Ritchison 2005, Jones *et al.* 2010).

Median clutch initiation date for grasshopper sparrows in Montana was June 16th (range: May 18th to July 17th) (Jones *et al.* 2010). Clutch size varies from three to six eggs, each of which has reddish-brown spots (Semenchuk 1992). Hovick *et al.* (2012) reported average clutch sizes of 2.42 eggs and 3.92 eggs for parasitized and non-parasitized nests, respectively. Jones *et al.* (2010) reported average clutch sizes of 4.3 ± 0.24, average number of nestlings of 3.9 ± 0.28 and 3.6 ± 0.33 fledglings per successful nest for grasshopper sparrows in Montana. Broods of three to five young have been reported elsewhere (Adler and Ritchison 2011). Incubation lasts 11 to 12 days in Alberta (Semenchuk 1992) and 10.9 ± 0.14 days in Montana (Jones *et al.* 2010). Both parents take turns feeding their altricial young. Young birds leave the nest after nine days in Alberta (Semenchuk 1992) and after 9.7 ± 0.17 days in Montana (Jones *et al.* 2010). In Nebraska, the majority of grasshopper sparrow nestlings hatch and fledge in June (Skipper and Kim 2013). Hovick *et al.* (2011) reported low post-fledgling survival (79% mortality) of grasshopper sparrows in Iowa, with the majority deaths a result of predators (48%) and exposure (28%). On average, fledglings begin flying four days after leaving the nest (range: 1 to 7) (Hovick *et al.* 2011). Age appears to have the greatest effect on survival (*i.e.*, as grasshopper fledglings age, their probability of survival increases) (Hovick *et al.* 2011). Ahlering *et al.* (2009) found that grasshopper densities in Saskatchewan and North Dakota increased in wet springs preceded by warmer winters.

5.2.1 Diet

In summer, grasshopper sparrows feed primarily on grasshoppers, whereas in winter they feed mainly on graminoid seeds, particularly panic grass (*Panicum* spp.) and sedges (*Carex* spp.) (Vickery 1996). Adler and Ritchison (2011) observed that 68.1% of prey items delivered by male and female grasshopper sparrows to their young were grasshoppers (Orthoptera). Other prey items noted by these authors included larvae (22.6%), crickets (Orthoptera) (5.6%), moths (Lepidoptera) (1.9%), bees (Hymenoptera), beetles (Coleoptera), and spiders (Araneae). Diets of grasshopper sparrows in Nebraska generally consisted of insect larvae from the Lepidoptera (butterflies and moths) and Orthoptera (grasshoppers and crickets) families (Kaspari and Joern 1996, Skipper and Kim 2013). Skipper and Kim (2013) found that lepidopteran larvae were selected disproportionately compared to their availability in the environment and were preferred over orthopterans, whereas Kaspari and Joern (1996) observed a preference for orthopterans. In Alberta, grasshopper sparrows feed primarily on insects (particularly grasshoppers and beetles), spiders, snails and other small invertebrates as well as seeds (Semenchuk 1992).

5.2.2 Predators

Hovick *et al.* (2011) observed that predation was the leading cause of mortality among grasshopper sparrow fledglings in Iowa, accounting for 49% of all deaths. Similarly, Jones *et al.*

(2010) observed that 35% of grasshopper sparrow nests in their Montana study area were depredated, and that predation was the leading cause of nest failure. Predators identified by Hovick *et al.* (2011) to be mostly responsible for failed grasshopper sparrow nests included meso-predators (*e.g.*, common raccoons, striped skunks, red foxes and domestic cats [*Felis catus*]) (48%) and snakes (6%). Potential grasshopper sparrow predators identified in North Dakota included common raccoons, thirteen-lined ground squirrels, red-sided garter snakes (*Thamnophis sirtalis*), American badgers, hawks and striped skunks (Renfrew and Ribic 2003). Loggerhead shrikes (*Lanius ludovicianus*) have been observed to prey on grasshopper sparrows in the southern United States (Vickery 1996).

Nest parasitism by brown-headed cowbirds is also common for grasshopper sparrows (Shaffer *et al.* 2003). Parasitism rates ranged from 0% to 58% for grasshopper sparrows in various parts of its range (summarized by Shaffer *et al.* 2003). In Manitoba, parasitism rates ranged from 27% to 30% (De Smet 1992b, Davis 1994, Davis and Sealy 2000). Cowbird brood parasitism was reported from 24% to 28% of nests in Iowa, with parasitized nests having significantly lower clutch sizes compared to non-parasitized nests (Hovick *et al.* 2012).

5.3 Habitat Requirements

5.3.1 General

Grasshopper sparrows have been described as a native prairie-obligate species (*e.g.*, Ahlering *et al*. 2009). In the breeding season, grasshopper sparrows generally inhabit intermediate grassland habitat, preferring thicker, brushier sites in shortgrass prairie and southwestern grasslands and drier, sparser sites in lush tallgrass prairies and eastern grasslands (Vickery 1996). Grasshopper sparrows are especially common in sandhills habitat (Wershler *et al*. 1991).

Grasshopper sparrows will also breed in tame pastures (Wilson and Belcher 1998) and cropland (Best *et al.* 1997). Davis and Duncan (1999) found that grasshopper sparrows were more frequent in pure crested wheat grass and wheat grass – grass (smooth brome, bluegrass) fields than in native mixedgrass prairie, and that the species was more frequent in wheat grass fields than wheat grass/alfalfa fields. McMaster and Davis (1998) found that grasshopper sparrows were more common in Permanent Cover Program (PCP) pastures than in cropland, and that frequency of occurrence for the species was higher in grazed PCP pastures than in hayed PCP pastures. Sutter and Brigham (1998) found no significant difference in grasshopper sparrow abundance between lightly grazed native prairie fields and lightly grazed crested wheat grass fields in southern Saskatchewan.

In Alberta, grasshopper sparrows prefer a mixture of lush grasses and low, open shrubland (Semenchuk 1992). Davis and Duncan (1999) found that grasshopper sparrow occurrence in Saskatchewan was positively correlated with vegetation height, crested wheat grass, northern wheat grass and needle grasses (*Hesperostipa* spp.). In the Milk River Watershed of southwestern Saskatchewan, grasshopper sparrow abundance was positively associated with vegetation volume and shrub cover (Henderson and Davis 2014).

Schneider (1998) provided a detailed description of grasshopper sparrow habitat characteristics in mixedgrass prairie in North Dakota. There, abundance of grasshopper sparrows was positively associated with percent grass cover, litter depth, vegetation height/density (visual obstruction), and density of low-growing shrubs (including buckbrush [*Symphoricarpos occidentalis*] and silverberry [*Elaeagnus commutata*]). Grasshopper sparrow abundance was also associated with plant communities dominated by shrubs and non-native grass species (*e.g.*, smooth brome, Kentucky bluegrass and quack grass) as well as plant communities dominated by Kentucky bluegrass and native grasses (*e.g.*, *Hesperostipa*, *Bouteloua*, *Koeleria* and *Schizachyrium*) (Schneider 1998). Schneider (1998) also found that grasshopper sparrow abundance was negatively associated with percent cover of prairie selaginella (*Selaginella densa*) and with plant communities dominated solely by native grasses. The strongest predictors of the presence of grasshopper sparrows were decreased cover of bare ground and prairie selaginella and increased litter cover (Schneider 1998).

In Wisconsin, grasshopper sparrows prefer dry prairie grassland (Sample and Hoffman 1989). In West Virginia, Whitmore (1981) observed that grasshopper sparrows preferred open grassland dominated by bunch grasses as opposed to grassland dominated by sod-forming species. In Florida, grasshopper sparrows preferred treeless, relatively poorly drained prairie grassland dominated by dwarf live oak (*Quercus minima*), saw palmetto (*Serenoa repens*) and various grasses and forbs (*e.g.*, pineland threeawn [*Aristida stricta*], bluestems [*Andropogon* spp.] and flat-topped goldenrod [*Euthamia minor*]) as well as tame pastures in various stages of regeneration (Delany *et al.* 1985). In a study covering portions of Colorado, Kansas, Montana, Nebraska, Oklahoma, South Dakota, Texas, Wisconsin and Wyoming, grasshopper sparrow abundance was negatively correlated with percent bare ground cover, amount of variation in litter depth and the amount of variation in forb and shrub heights (Rotenberry and Wiens 1980). Grasshopper sparrow abundance in this study was positively correlated with percent grass cover, percent litter cover, effective vegetation height and density as well as litter depth (Rotenberry and Wiens 1980). In tallgrass prairie in Kansas, grasshopper sparrow abundance was positively associated with forb cover and negatively associated with warm-season grass cover and visual obstruction (Johnson and Sandercock 2010).

On a regional scale, annual variability in climate may influence habitat selection by grasshopper sparrows in the Northern Great Plains (Ahlering *et al.* 2015). Ahlering *et al.* (2015) observed higher densities of grasshopper sparrows in areas that received higher spring precipitation. The authors suggest that high spring moisture may signal to arriving grasshopper sparrows that these sites have higher vegetation and invertebrate prey productivity.

5.3.2 Nesting Habitat

No information on grasshopper sparrow nesting habitat in Alberta was found in the literature. In Montana, grasshopper sparrows appeared to select for vertical vegetative characteristics at the exact nest site as opposed to habitat characteristics in the immediate vicinity (*i.e.*, 5 m radius) (Dieni and Jones 2003). In this study, grasshopper sparrows generally avoided western wheat grass as a nesting substrate, opting for the shorter-statured species blue grama. Grasshopper sparrows also appeared to select for low bare soil cover in the immediate vicinity of the nest (Dieni and Jones 2003).

In Kentucky, grasshopper sparrows selected nest sites that had shorter, denser vegetation, fewer shrubs and more trees than surrounding areas (from 0.5 m to 1 m) (Sutter and Ritchison 2005). Similarly, Whitmore (1979) suggested that selection of nest sites by female grasshopper sparrows was likely influenced by the density of ground cover, with vegetation in the vicinity of the nest permitting movement while providing adequate concealment from potential predators. In tallgrass prairie in Iowa, Hovick *et al.* (2012) observed higher nest survival rates when nests were located in sites with lower cover of cool season grasses and lower total vegetation cover within 5 m of the nest. The authors speculate that higher vegetation cover around the nest may provide refugia for predators such as snakes and decrease the available foraging habitat (see discussion below). Also in Iowa, Patterson and Best (1996) observed lower grasshopper sparrow nest survival in fields with higher vertical vegetation cover and found that grasshopper sparrows were most abundant in fields with moderate vegetation height.

5.3.3 Foraging Habitat

Grasshopper sparrows are exclusively ground foragers (Vickery 1996). Whitmore (1981) found that grasshopper sparrow territories in West Virginia had sparser, shorter vegetation with more bare ground, fewer shrubs and lower litter cover compared to non-territories. Ahlering *et al.* (2015) observed an inverse relationship between grasshopper sparrow densities and percent shrub cover in Saskatchewan and North Dakota. Although grassland habitat with dense shrub cover is generally avoided by the grasshopper sparrow, a baseline level of shrub cover is preferred by birds in the western part of its range (Vickery 1996).

Small *et al.* (2015) monitored post-fledgling movements of dependent (*i.e.*, less than 32 days old and independent (*i.e.*, greater than 32 days old) juvenile grasshopper sparrows in Maryland. Dependent birds favoured habitats with higher bare ground and litter cover and greater plant species richness compared to random plots, whereas independent juveniles favoured habitat with high bare ground cover only. Dependent juveniles were found more often in habitat with higher total vegetation and warm-season grass cover and lower bare ground and forb cover than independent birds. The authors suggest that dependent juveniles are less able to escape from predators because their flight feathers are not fully developed and thus must remain in denser vegetation, whereas independent birds are better able to escape from predators and forage in more open areas.

5.3.4 Area Requirements

Although individual grasshopper sparrow territories are actually quite small (*i.e.*, less than 2 ha), the species prefers large, contiguous blocks of native grassland (Vickery 1996, Dechant *et al.* 2002e). For example, Delany *et al.* (1985) observed territories of 1.8 ± 0.96 ha in Florida, and Wiens (1969) observed territories of 0.85 ha in Wisconsin. In Nebraska, the minimum area in which grasshopper sparrows were found was 8 ha to 12 ha, with a perimeter-area ratio of 0.018 (Helzer and Jelinski 1999). Minimum areas were even larger in Illinois (*i.e.*, 10 ha to 30 ha for grasshopper sparrow occurrence and ≥30 ha for a breeding population) (Herkert 1991, 1994).

With respect to fledglings, Hovick *et al*. (2011) observed average daily movements from the nest site of 37.5 ± 3.1 m. The maximum recorded distance travelled by a fledgling was 135 m. Small

et al. (2015) observed average daily movements of recently fledged dependent juveniles from nests of 88.8 m (range: 3.4 m to 1,042 m). Dependent and independent birds moved an average of 46.3 m (range: 6.1 m to 401.5 m) and 146.1 m (range: 5.2 m to 966 m) between daily locations.

Grasshopper sparrows are area sensitive, being found in higher abundances in larger patches of mixedgrass prairie in Saskatchewan (Davis 2004). Davis (2004) found that grasshopper sparrows preferred patches at least 134 ha in size.

6 CHESTNUT-COLLARED LONGSPUR

6.1 Background

The chestnut-collared longspur is a medium-sized songbird with a chestnut-coloured collar, black underparts and a white or buff-coloured face (Semenchuk 1992). The name 'longspur' is a reference to the long, slender claw on the hind toe (Bleho *et al.* 2015). The chestnut-collared longspur can be distinguished from other longspurs by the black triangular patch in the centre of the tail and its white outer tail feathers (Bleho *et al.* 2015). Chestnut-collared longspurs are native grassland specialists, occurring throughout the shortgrass and mixedgrass prairies of the Great Plains (COSEWIC 2009). In Canada, the chestnut-collared longspur breeds in dry mixedgrass and mixedgrass prairies in southeastern Alberta, southern Saskatchewan and southwestern Manitoba (COSEWIC 2009). Its extent of occurrence in this country is estimated at 292,000 km^2 (COSEWIC 2009). In Alberta, the chestnut-collared longspur is almost entirely confined to the Grassland Natural Region, and it attains its highest densities in the central portion of the subregion (NRC 2006, Federation of Alberta Naturalists 2007). Chestnut-collared longspurs over-winter in the southern United States and northern Mexico (Bleho *et al.* 2015).

The chestnut-collared longspur is currently listed as 'At Risk' under the Alberta General status listing (GoA 2016). The species is listed as a 'non-game animal' under the provincial *Wildlife Act*. Chestnut-collared longspurs and their eggs and nests are also protected from harm under the *Migratory Birds Convention Act*. The chestnut-collared longspur was assessed by COSEWIC in 2009 and given a status of 'Threatened', a designation the species currently retains (COSEWIC 2009, 2016). Based on the recommendation by COSEWIC, the chestnut-collared longspur was added to *SARA* in 2012 with a listing of 'Threatened'. Internationally, the chestnut-collared longspur has been designated as a 'Near Threatened' species by the IUCN (BirdLife International 2012f).

AEP has yet to prepare a formal status report for the chestnut-collared longspur. A proposed Recovery Strategy for the chestnut-collared longspur was released by the federal government in early 2016 (Environment and Climate Change Canada 2016).

6.2 Ecology

Chestnut-collared longspurs arrive on their breeding grounds in southern Alberta in mid- to late April (Semenchuk 1992). Breeding occurs from April to August in Alberta (Federation of Alberta Naturalists 2007). Chestnut-collared longspurs begin returning to their wintering grounds

in August and September (Semenchuk 1992). Nests are built on the ground by the females using grasses and other vegetation (Semenchuk 1992).

Clutch sizes range from three to six eggs (Semenchuk 1992). Harris (1944) observed an average clutch size (n=10) of 4.8 eggs in Manitoba, while Lloyd and Martin (2006) observed an average clutch size of 4.0 eggs in Montana. Incubation lasts 10 to 12 days, and nestlings fledge after nine to 11 days (Harris 1944, Semenchuk 1992). Incubation is performed strictly by the female (Harris 1944). Young chestnut-collared longspurs are able to fly a few days after fledging, and are completely independent approximately 24 days after hatching (Semenchuk 1992). The chestnut-collared longspur frequently has double broods, with the male taking over nest-tending when the female begins a second brood (Semenchuk 1992). Pairs may attempt up to four clutches in a single breeding season if preceding clutches fail (Bleho *et al.* 2015). Both males and females tend young broods (Semenchuk 1992). Nesting success rates (*i.e.*, the number of nests producing at least one young) reported for Alberta, Saskatchewan and Manitoba are 55.9%, 41% and 45%, respectively (Pipher 2011, Bleho *et al.* 2015).

Nest predation has been identified as the primary factor responsible for reproductive failure in grassland birds in general (Best *et al.* 1997, Koford 1999, Jones *et al.* 2010) and for chestnut-collared longspurs in particular (Lloyd and Martin 2006, Lusk 2009, Pipher 2011). Pipher (2011) and Lusk (2011) observed predation rates of 50% and 87%, respectively, for chestnut-collared longspur nests in Grasslands National Park in Saskatchewan, whereas in Montana, Lloyd and Martin (2006) noted predation rates of 41.9% and 49.3% for nests in native prairie and tame pasture, respectively. Maher (1973) reported that nest predation at Matador, Saskatchewan accounted for 97% and 72% of egg and nestling mortalities, respectively.

6.2.1 Diet

Adult chestnut-collared longspurs forage on the ground, picking up insects and other small invertebrates as well as seeds during the breeding season (Semenchuk 1992). The species feeds primarily on seeds during the winter (Semenchuk 1992). Young chestnut-collared longspurs are fed insects, primarily grasshoppers (Semenchuk 1992).

In South Dakota, stomach contents of adult chestnut-collared longspurs (n=17) contained primarily insects, including 60% to 95% Orthoptera (grasshoppers and crickets), 0% to 10% Coleoptera (beetles), 0% to 11% Lepidoptera (butterflies and moths) and less than 2% each of Hymenoptera (ants and bees), Hemiptera/Homoptera (true bugs), Diptera (true flies) and Aranae (spiders) (Wiens and Rotenberry 1979). Adult chestnut-collared longspur diets also contained 0% to 23% grass seeds.

In Saskatchewan, nestling chestnut-collared longspur diets varied throughout the summer season (*i.e.*, May to August) and consisted primarily of insects (Maher 1979). Important prey items included Orthoptera (30% to 66%), Lepidoptera (9% to 27%), Homoptera/Hemiptera (9% to 30%), Aranae (3% to 8%), Hymenoptera (1% to 9%), Coleoptera (0% to 3%), Hemiptera (1% to 6%) and Diptera (0% to 2%), among others.

6.2.2 Predators

Known predators of adult and fledgling chestnut-collared longspurs include swift foxes (*Vulpes velox*), thirteen-lined ground squirrels and burrowing owls (*Athene cunicularia*) (summarized by Bleho *et al.* 2015). Suspected chestnut-collared longspur predators include coyotes, red foxes, American badgers, common raccoons, striped skunks, merlins, northern harrier, American kestrel, prairie falcon and loggerhead shrikes (Bleho *et al.* 2015). Kirkham and Davis (2013) recorded nest predation by Richardson's ground squirrels in southwestern Saskatchewan. Suspected nest predators include deer (*Odocoileus* spp.), coyotes, striped skunks, black-tailed prairie dogs (*Cynomys ludovicianus*), deer mice (*Peromyscus maniculatus*), meadow voles (*Microtus pennsylvanicus*), meadow jumping mice (*Zapus hudsonius*), harvest mice (*Reithrodontomys* spp.), prairie rattlesnakes (*Crotalus viridis*), bullsnakes, gulls (*Larus* spp.), hawks (*Buteo* spp.), short-eared owls (*Asio flammeus*), loggerhead shrikes, American crows, black-billed magpies and western meadowlarks (*Sturnella neglecta*) (summarized by Bleho *et al.* 2015).

Chestnut-collared longspurs appear to be infrequently parasitized by brown-headed cowbirds (Bleho *et al.* 2015). For example, Davis and Sealy (2000) found that chestnut-collared longspurs were parasitized 2 to 3 times less frequently than most other songbirds in southern Manitoba. Overall, cowbird parasitism rates reported in the literature for chestnut-collared longspurs ranged from 0% to 25% (summarized by Shaffer *et al.* 2003). The low parasitism rate for chestnut-collared longspurs is not a result of egg rejection behaviour, but may be a function of anti-parasite strategies such as aggressive nest defense behaviour (Davis *et al.* 2002).

6.3 Habitat Requirements

6.3.1 General

Chestnut-collared longspurs are native prairie obligates (Semenchuk 1992, COSEWIC 2009). In Alberta, the species is restricted to the Grassland Natural Region (Federation of Alberta Naturalists 2007). Wershler *et al*. (1991) described chestnut-collared longspurs as common in the mixedgrass prairie and local in the northern fescue prairie. Typical chestnut-collared longspur habitat consists of mixedgrass or shortgrass prairie on flat to rolling topography (Bleho *et al.* 2015).

Habitat occupied by chestnut-collared longspurs usually has grassy vegetation less than 20 cm to 30 cm tall and minimal litter cover (Bleho *et al.* 2015). Similar to some other grassland birds, habitats with scattered shrubs are preferred, with the shrubs used as perching sites, and habitats with extensive shrub cover are avoided (Arnold and Higgins 1986, Federation of Alberta Naturalists 2007). Henderson and Davis (2014) observed that abundances of chestnut-collared longspurs increased with higher amounts of bare soil and decreased as vegetation and litter cover increased. Davis (2004) found higher abundances of chestnut-collared longspurs in native mixedgrass prairie fields in Saskatchewan that had lower densities of tall grass and standing dead vegetation. Anstey *et al.* (1995) found that chestnut-collared longspurs in Saskatchewan preferred native prairie habitat with low vegetation cover and low litter cover. Dale (1983, 1984) observed that mixedgrass prairie habitats occupied by chestnut-collared longspurs in Saskatchewan had lower litter, dwarf shrub and grass cover, lower forb heights, lower vertical

density and higher bare ground cover than unoccupied sites. Occupied sites had, on average, 83.3% litter cover, 3.1% dwarf shrub cover, 38.5% grass cover and 11.5% bare ground cover (Dale 1983, 1984). Davis *et al.* (1999) found that chestnut-collared longspur occurrence in native mixedgrass prairie was negatively associated with litter depth and density of narrow-leaved grasses less than 10 cm tall. In contrast, Davis and Duncan (1999) observed that chestnut-collared longspurs in Saskatchewan preferred native mixedgrass prairie to tame pasture, with abundance positively correlated with cover of June grass (*Koeleria macrantha*) and prairie selaginella.

In Saskatchewan, chestnut-collared longspurs occurred as frequently in native mixedgrass prairie as tame pastures, but were more common in tame pastures than either hay fields or cropland (Davis *et al*. 1999). Similarly, Sutter and Brigham (1998) observed no significant difference in abundances of chestnut-collared longspurs in lightly grazed mixedgrass prairie and lightly grazed tame pasture dominated by crested wheat grass. In the Hand Hills of Alberta, chestnut-collared longspurs were found to avoid cropland, including fallow, seeded and recently cultivated lands (Owens and Myres 1973). McMaster and Davis (1998) found that chestnut-collared longspurs in the Prairie Provinces were more common in PCP fields than cropland, and that frequency of occurrence was higher in grazed PCP fields compared to hayed PCP fields. Chestnut-collared longspurs are found in cropland in Alberta, preferring minimum-till fields compared to conventionally-tilled fields, especially summer fallow and spring cereal stands (as opposed to winter wheat) (Martin and Forsyth 2003).

6.3.2 Nesting Habitat

In Montana, Lloyd and Martin (2005) found that chestnut-collared longspurs did not discriminate between native prairie habitats and crested wheat grass tame pasture for nesting. However, chestnut-collared longspurs had reduced fitness (reproductive success), including lower daily nest survival rates, slower growth rates for nestlings and smaller nestling body masses, in tame pasture compared to native prairie habitat. Although chestnut-collared longspurs were originally thought to avoid nesting in cultivated fields (Semenchuk 1992), more recent evidence suggests that the species will breed in cropland (Martin and Forsyth 2003).

In Manitoba, territories were located in native prairie habitats with short, sparse grass cover and variable quantities of buckbrush shrubs, which are used as perches by singing males (Harris 1944). Nests are placed on the ground in light to moderately thick grass cover, sometimes in scattered growths of low-growing buckbrush (Harris 1944).

On a micro-habitat scale, Davis (2005) found that chestnut-collared longspurs in Saskatchewan mixedgrass prairie nested in sites characterized by tall vegetation, a high density of dead vegetation within 30 cm of the ground surface, high litter cover and low cover of bare soil. On a slightly smaller scale, nest sites were located in areas with shorter and sparser vegetation. Therefore, chestnut-collared longspurs nested in areas with shorter and sparser vegetation compared to other grassland passerines (*e.g.*, Sprague's pipit), but selected nesting sites with taller, denser vegetation within these areas (Davis 2005). Similarly, Dieni and Jones (2003) observed that chestnut-collared longspurs in Montana used nesting sites with taller, denser vegetation were generally available. These authors also observed a positive association between

chestnut-collared longspur nests and cover of prairie selaginella and a negative association between nests and western wheat grass.

In Grasslands National Park in Saskatchewan, chestnut-collared longspurs, like all of the six songbird species studied, chose nest sites with significantly greater vegetation height and density, greater litter depth and sites with significantly less biocrust (*i.e.*, moss and lichen) or bare soil cover, compared with unused microhabitats (Pipher 2011). Similarly, Lusk (2009) and Lusk and Koper (2013), also working in Grasslands National Park, found that chestnut-collared longspurs selected nesting sites with denser, taller vegetation than was generally available in the surrounding habitat. In comparison to other songbirds (*e.g.*, Sprague's pipit), however, chestnut-collared longspurs select nesting sites with shorter vegetation, lower vegetation cover, higher forb cover, higher cover of biocrusts and lower litter cover (Pipher 2011).

Chestnut-collared longspurs nests are often situated next to cow dung or other visible landmarks, such as large grass clumps (Harris 1944, Semenchuk 1992, Davis 2005, Pipher 2011). In Alberta, nests are placed on scrapes in the ground in low and slightly moist grassland with light to moderately thick vegetation (Semenchuk 1992).

6.3.3 Foraging Habitat

Chestnut-collared longspurs forage on or near the ground in open habitat (Bleho *et al.* 2015). Little information specifically dealing with chestnut-collared longspur foraging habitat appears to have been published in the literature.

6.3.4 Area Requirements

In southeastern Alberta, mean chestnut-collared longspur territory size is 1 ha (range: 0.25 ha to 4.0 ha) (Fairfield 1968, cited in Bleho *et al*. 2015). In Saskatchewan, the typical territory size is 0.4 ha to 0.8 ha, with a maximum recorded size of 4 ha (Fairfield 1968, cited in Bleho *et al*. 2015). In Montana, Lloyd and Martin (2005) observed chestnut-collared longspur densities of 1.8 pairs/ha and 1.2 pairs/ha in native prairie and tame pasture habitats, respectively. In southeastern Alberta, chestnut-collared longspur densities of 1.1 pairs/ha to 1.4 pairs/ha have been reported (Bleho *et al.* 2015). At Matador, Saskatchewan, densities of 0.7 pairs/ha to 1.2 pairs/ha were observed on grazed plots and 0 pairs/ha to 0.2 pairs/ha were observed on ungrazed plots (Bleho *et al.* 2015). Also at Matador, Maher (1979) observed breeding densities of 0.09 pairs/ha on ungrazed plots and 0.9 pairs/ha on grazed plots. Harris (1944) noted that chestnut-collared longspur territories in Manitoba were roughly circular in shape with the nest located in the middle. Two territories measured by this author were approximately 0.2 ha and 0.4 ha in size.

Chestnut-collared longspurs are area sensitive, being found in higher abundance as patch size increases (Davis 2004). In mixedgrass prairie in Saskatchewan, Davis (2004) found that chestnut-collared longspurs preferred patches at least 39 ha in size.

7 GRAZING AND GRASSLAND BIRD GROUP 2

Endemic grassland birds evolved in a dynamic landscape influenced by fire, drought and vast herds of grazing bison and other wild herbivores. These factors created a diversity of habitat conditions, stimulating a range of adaptations among birds to differing degrees of disturbance. Correspondingly, grassland birds can be arranged along a gradient of increasing vegetation cover and decreasing tolerance to disturbance (Knopf 1996a) (Figure III-1). At one extreme of the spectrum are species like the mountain plover (*Charadrius montanus*) that favour extremely short vegetation and high amounts of grazing disturbance (see Section III-E). At the other end of the continuum are species like Sprague's pipits and Baird's sparrows, which prefer greater amounts of vegetation cover and decrease in abundance under heavy grazing. Long-billed curlews and upland sandpipers fall in between.

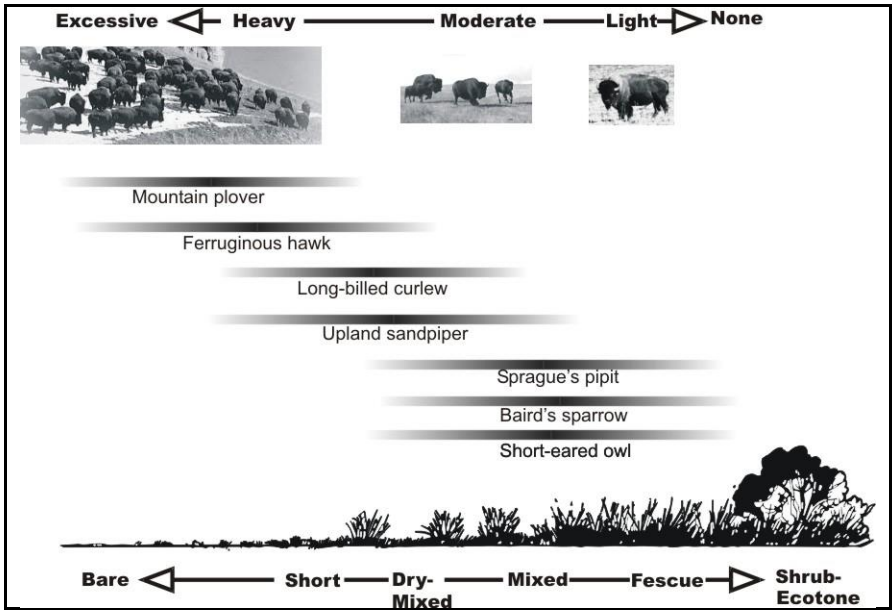

Figure III-1. Distribution of Grassland Birds in relation to Grassland Type and Historical Grazing Pressure (adapted from Knopf 1996a)

Grazing affects grassland habitat characteristics by influencing vegetation species composition and productivity and by changing horizontal and vertical vegetation structure (*e.g.*, height, density and litter cover) (Severson and Urness 1994). These characteristics influence the suitability of an area to birds by affecting key parameters, such as the availability of display perches, nesting and brood-rearing cover and prey diversity and abundance. To a lesser extent, large grazers may also impact grassland bird productivity due to direct trampling of nests.

The response of grassland birds to grazing pressure varies depending on soil type, moisture regime and vegetation type (Knopf 1996b). For example, species that usually occur in medium height mixedgrass prairie typically respond negatively to heavy grazing in arid grasslands but respond favourably to increased grazing intensity in mesic (*e.g.*, tallgrass) prairie. Therefore, although the habitat requirements of birds may exist naturally at one location, a differential

degree of disturbance may be required at other locations to establish similar structural or species composition parameters (Knopf 1996b). Even at the same location, birds may exhibit a varying response to grazing in relation to variation in seasonal precipitation. It is therefore important to set stocking rates and grazing frequency in accordance with local ecological conditions and to adjust stocking rates during drought conditions.

Livestock grazing can be used to create habitat conditions that are suitable for a diverse suite of grassland birds, such as those considered in this report. Appropriately managed, grazing can stimulate habitat heterogeneity, reduce dense accumulation of litter and reduce encroachment of woody species. These properties are considered beneficial to long-billed curlews, upland sandpipers, Sprague's pipits, Baird's sparrows, grasshopper sparrows and chestnut-collared longspurs. In the absence of grazing, other mechanisms, such as prescribed burns or periodic mowing, are required to create similar conditions. Section 9 includes recommendations for appropriate burning and mowing strategies to enhance habitat for long-billed curlews, upland sandpipers, Sprague's pipits, Baird's sparrows, grasshopper sparrows and chestnut-collared longspurs.

The following subsections discuss in greater depth the response of long-billed curlews, upland sandpipers, Sprague's pipits, Baird's sparrows, grasshopper sparrows and chestnut-collared longspurs to grazing. Overall, these species have similar habitat requirements and respond favourably to light to moderate grazing, depending on local site conditions. Unlike Sprague's pipits, Baird's sparrows and grasshopper sparrows, long-billed curlews, upland sandpipers and chestnut-collared longspurs require larger areas of shorter vegetation during the breeding season, making them more reliant on sustained patches of heavier grazing.

7.1 Long-billed Curlew Response to Grazing

Overall, grazing is considered beneficial for creating suitable habitat for long-billed curlews (Dechant *et al.* 2002a). The effects of grazing on long-billed curlew habitats vary depending on habitat conditions, including soil type, vegetation structure and moisture regime (De Smet 1992a, Hill 1998, Dechant *et al.* 2002a). Environment Canada's (2013) threat assessment lists overgrazing as a low level of concern. Based on a review of studies conducted in the Northern Great Plains, Kantrud and Kologiski (1982) found that long-billed curlews generally preferred lightly grazed aridic soils and heavily grazed moister soils. In Alberta, breeding densities of long-billed curlews are often highest in moderately grazed mixedgrass prairie with sandy loam soils (De Smet 1992a). Moderate livestock grazing is thought to be compatible with maintaining the variable vegetation structure required by long-billed curlews throughout the breeding season (Hill 1998). Moderate grazing usually promotes patchy vegetation structure, creating areas of shorter cover used by long-billed curlews for nesting and foraging, and retaining areas of denser, taller vegetation used during brood-rearing (Hill 1998, Dechant *et al.* 2002a). Patches of open ground and shorter grass are also important for predator detection and adult and chick mobility (Pampush and Anthony 1993, Dugger and Dugger 2002).

Although several studies have assessed long-billed curlew response to grazing in the Northern Great Plains region of the United States (Bicak *et al.* 1982, Kantrud and Kologiski 1982, Cochran and Anderson 1987, Bock *et al.* 1993), few studies have been conducted in the

Canadian prairies (Prescott *et al.* 1993, Prescott and Wagner 1996). In southern Alberta, Prescott and Wagner (1996) found that long-billed curlews were most common in continuously grazed and early-season (April to mid-June) grazed mixedgrass prairie. Long-billed curlews were present in only a few tame pasture sites and did not occur in areas where grazing was deferred until after mid-June, when pairs had already established territories (Prescott and Wagner 1996). Grazing systems that created reduced vegetation height and density early in the spring during the pre-laying and laying periods were similarly considered beneficial to long-billed curlews in Idaho (Bicak *et al.* 1982, Redmond and Jenni 1982). Removal of tall, dense residual vegetation before the pre-laying period allowed adults to stay in their territory to forage (Redmond and Jenni 1986).

Livestock grazing not only has the potential to alter vegetation structure, but may also impact long-billed curlew nesting success. In Wyoming, Cochran and Anderson (1987) found that nests in areas that were grazed during the nesting period had lower hatching success than nests in ungrazed areas. Redmond and Jenni (1986) noted that of 119 nests in western Idaho, 4.2% of nests were abandoned or lost due to livestock disturbance. In Nevada, trampling by livestock was the cause of 7.5% of nest failures (Hartman and Oring 2009). In general, livestock trampling of nests is not likely to be a significant concern unless stocking rates are extremely high during the nesting season (Dugger and Dugger 2002). Other factors that influence the likelihood of nest abandonment or destruction include the length of incubation prior to livestock introduction as well as the duration and frequency of livestock disturbance (De Smet 1992a).

7.2 Upland Sandpiper Response to Grazing

As with long-billed curlews, a grazing system that promotes patches of high, moderate and lighter use is also considered beneficial for meeting the diverse habitat requirements of upland sandpipers. Shorter vegetation in more heavily grazed areas provide upland sandpipers with suitable foraging sites, while patches with intermediate grazing intensity and short to medium vegetation may be suitable for brood-rearing. Less heavily used areas with taller vegetation may provide suitable cover for nesting (Dechant *et al.* 2002b). Like long-billed curlews, upland sandpipers vary in their response to grazing in different regions depending on vegetation type, moisture regime and soil conditions (Saab *et al.* 1995). Upland sandpipers responded positively to moderate grazing in areas with moister soils in the Northern Great Plains (Kantrud and Kologiski 1982) and to moderate and heavy grazing in tallgrass prairie in Missouri (Skinner 1975). In mixedgrass prairie in Saskatchewan, Dale (1984) found that upland sandpipers nested only in grazed fields and were not observed in ungrazed fields. Upland sandpipers were only observed on deferred-grazed (grazed after July 15) mixedgrass prairie in southern Alberta (Prescott and Wagner 1996). None were present on continuous or early-season grazed prairie or in tame pastures (Prescott and Wagner 1996).

Several studies have examined the effects of grazing on nest density or nesting success of upland sandpipers in mixedgrass prairie in south-central North Dakota (Kirsh and Higgins 1976, Messmer 1990, Bowen and Kruse 1993, Sedivec 1994). Bowen and Kruse (1993) found that upland sandpiper nest densities were lower in fields that were grazed during the nesting season (*i.e.*, season long, spring or fall-and-spring grazing) than in ungrazed fields or fields grazed in the fall. In central Wisconsin, Ailes (1980) reported that upland sandpipers did not nest in areas of

heavy livestock grazing. Livestock trampling was identified as the primary cause of nest destruction in this study (Ailes 1980). Similarly, Kirsh and Higgins (1976) found that upland sandpiper nest success was lower on grazed land (48%) than on undisturbed grassland (71%) or previously burned grassland. Nesting success was also lower on grazed fields than on idle fields in the prairie pothole region of South Dakota, North Dakota, Montana and Manitoba (Kantrud and Higgins 1992). However, of 32% nest losses on grazed fields, only 1% was directly due to livestock disturbance (Kantrud and Higgins 1992). Contrary to the former studies, Messmer (1990) found that upland sandpiper nest density and nest success were higher under twice-over deferred and season-long grazing systems than on idle (ungrazed) fields. Under the twice-over deferred rotation system, fields were grazed twice per season with a two-month rest in grazing (Messmer 1990). Of interest, Messmer (1990) noted that the average density of breeding sandpipers was highest on the short-duration grazing system where fields were sequentially grazed for a week and then rested for a month from late May or early June until October (Messmer 1990). Sedivec (1994) similarly reported that upland sandpiper nesting density was significantly higher on grazed mixedgrass fields than in idle grasslands.

7.3 Sprague's Pipit and Baird's Sparrow Response to Grazing

Sprague's pipits and Baird's sparrows appear to favour light to moderate grazing intensity throughout much of their range (Dechant *et al.* 2002c,d). As with the previous species, the effects of grazing intensity vary according to local environmental conditions (Bock *et al.* 1993, Saab *et al.* 1995, Robbins and Dale 1999). Moderate grazing may be tolerated by these species in mesic grasslands; however, lighter use is considered more appropriate in arid mixedgrass and dry mixedgrass prairies (Prescott 1997, Robbins and Dale 1999, Dechant *et al.* 2002d). Periodic grazing or other types of disturbances such as burning or mowing are considered particularly beneficial to Sprague's pipits and Baird's sparrows in areas with uniformly tall, dense vegetation and excessive litter accumulation (Madden 1996, Madden *et al.* 2000, Dechant *et al.* 2002c.d). Both species respond favourably to intermediate vegetation height and density and tend to avoid areas with dense, matted vegetation or litter build-up exceeding 4 cm (Dechant *et al.* 2002c,d).

Several studies in fescue and mixedgrass prairie in Alberta and Saskatchewan have documented the similar tendency for Sprague's pipits and Baird's sparrows to respond negatively to heavy grazing and to occur more frequently in ungrazed or light to moderately grazed areas (Owens and Myres 1973, Maher 1979, Dale 1984, Anstey *et al.* 1995, Koper and Schmiegelow 2015). For example, Koper and Schmiegelow (2006), working in the dry mixedgrass prairie in southern Alberta, suggested that both Baird's sparrows and Sprague's pipits were relatively insensitive to light and moderate grazing intensities. Similarly, in Grasslands National Park in Saskatchewan, Sliwinski (2011) observed that Sprague's pipits and Baird's sparrows decreased linearly in relative abundance as stocking rates increased from ungrazed controls to very heavily grazed fields and Sliwinski and Koper (2015) observed that relative abundance of Baird's sparrows decreased as stocking rates increased. Sprague's pipits showed more ambiguous results, appearing to decrease with increasing stocking rates in year 1, but showing no significant effect after two years of grazing (Sliwinski and Koper 2015). Sprague's pipits and Baird's sparrows were associated with higher range condition in the Grassland Ecoregion of southern Saskatchewan (Anstey *et al.* 1995), and Sprague's pipits were significantly more common in higher condition rangeland with low shrub cover in fescue grasslands of the Cypress Hills (Hull

2002). Similar results to those found in Canada have been reported for the Northern Great Plains states, with Baird's sparrow preferring light grazing intensity and Sprague's pipits preferring light to moderate grazing intensity (Kantrud and Kologiski 1982, 1983). One exception to these general conclusions was a study in North Dakota, which found that Sprague's pipits preferred both heavily and moderately grazed plots over lightly grazed plots in an area with higher grassland productivity (Kantrud 1981).

A comparison of various grazing systems in mixedgrass prairie of southern Alberta found that Sprague's pipits were most common in early-season grazed native prairie (grazed before July 15) in 1993 and 1995 (Prescott *et al.* 1993, Prescott and Wagner 1996). Sprague's pipits varied in their response to continuous and deferred grazing (grazed after July 15) between years, but were consistently more common in managed native prairie than in tame pastures (Prescott *et al.* 1993, Prescott and Wagner 1996). Baird's sparrows showed a preference for early-season grazed and deferred grazed prairie in 1993; however, they occurred with similar frequencies in all grazing treatments in 1995 (including tame pasture) (Prescott *et al.* 1993, Prescott and Wagner 1996). High spring rainfall and improved plant vigour in all treatments may have influenced the greater distribution of Baird's sparrows in 1995 (Prescott and Wagner 1996).

Few studies have examined the effects of various grazing systems or grazing intensities on the reproductive success of either Sprague's pipits or Baird's sparrows. Only one study was found in the literature; in Grasslands National Park, Pipher (2011) found no significant effect of livestock grazing on Baird's sparrow nesting success, whereas Sprague's pipits exhibited a non-linear response to livestock grazing, with nesting success highest in ungrazed and heavily grazed fields.

7.4 Grasshopper Sparrow Response to Grazing

Livestock grazing appears to be conducive to maintaining grasshopper sparrow habitat. In North Dakota, grazed fields were favoured by grasshopper sparrows over idled fields (Wiens 1973). Grazing was a common feature of all grasshopper sparrow territories observed by Delany *et al.* (1985) in Florida. Grasshopper sparrows appear to prefer moderately disturbed grassland habitats (Whitmore 1981). Kantrud (1981) observed higher numbers of grasshopper sparrows in lightly and moderately grazed native prairie fields compared to heavily grazed native prairie fields. The author observed similar numbers of grasshopper sparrows in heavily grazed and recently hayed native grassland fields. Kantrud and Kologiski (1982) found significantly greater grasshopper sparrow densities on lightly grazed plots compared to heavily grazed plots, with moderately grazed plots supporting intermediate densities of sparrows. Similarly, Arnold and Higgins (1986) observed higher densities of grasshopper sparrows in North Dakota in lightly grazed fields. Fritcher *et al.* (2004) observed higher grasshopper sparrow densities on later seral stage native mixedgrass prairie in South Dakota, with sparrows nearly absent from early seral stage plant communities. Later seral stages were dominated by western wheat grass and green needle grass (*Nassella viridula*). In Grasslands National Park in Saskatchewan, Sliwinski (2011) observed that grasshopper sparrow abundance was highest at moderate and light stocking rates and declined at high stocking rates (*i.e.*, above 0.40 AUMs/ha).

Vegetation height, which is related to grazing intensity, is important for grasshopper sparrows. Rotenberry and Wiens (1980) found that grasshopper sparrows were positively correlated with

vegetation height and density in mixedgrass prairie. Patterson and Best (1996) found grasshopper sparrows in Iowa to be most abundant in fields with moderate vegetation height. Similarly, Hovick *et al.* (2012) found that vegetation cover near the nest significantly affected nest survival, with grasshopper sparrows preferring lower vegetation cover for nesting, and that vegetation cover was a direct result of stocking rates. These authors recommend stocking at a level that reduces overall biomass to the point where obvious grazing lawns are created but where areas suitable for nesting remain. Livestock grazing may be a useful tool to thin dense vegetation or decrease vegetation height to improve habitat for grasshopper sparrows (Kantrud 1981, Whitmore 1981).

Patch-burn grazing (also known as pyric herbivory) has been recommended for grassland bird habitat management in the tallgrass prairie ecosystem and is designed to create habitat heterogeneity by mimicking evolutionary grazing patterns (Fuhlendorf and Engle 2001, Fuhlendorf *et al.* 2006). Fuhlendorf *et al.* (2006) reported higher grassland bird species diversity in a large contiguous block of tallgrass prairie in the Flint Hills of Oklahoma that had been patch-burned and grazed compared to more traditional management (*i.e.*, annual spring burns followed by grazing from mid-April to mid-July). In their study of grazing and prescribed burning in much smaller tallgrass prairie fields in Iowa, Hovick *et al*. (2012) observed higher grasshopper nest survival in patch-burn grazed fields (*i.e.*, fields grazed from May to September, with prescribed burning occurring in a portion of the fields each spring) compared to grazed-and-burned fields (*i.e.*, fields that were grazed annually from May to September and then burned after three years to remove shrubs), although the difference was not significant. The authors suggest that either grazing treatment could be used to provide suitable habitat for nesting grasshopper sparrows.

While grazing appears to be preferred over non-use, some exceptions occur. For example, Bock *et al.* (1984) observed significantly higher numbers of grasshopper sparrows in both winter and summer in Arizona native prairie that had been rested for 12 to 13 years compared to grassland that had been continuously grazed on an annual basis. These results may be explained by the climate of this semi-desert grassland ecosystem, with ungrazed sites potentially having higher bare ground than, for example, mixedgrass prairie without the influence of livestock grazing. Sutter and Ritchison (2005) observed significantly higher nest success rates for grasshopper sparrows nesting in ungrazed (70%) versus grazed (25%) fields in Kentucky. Similarly, the authors found that clutch sizes were significantly higher in ungrazed fields (4.48) compared to fields grazed by livestock (3.91). In terms of vegetation, grazed sites had lower litter, more shrubs and shorter, less dense vegetation than ungrazed sites (Sutter and Ritchison 2005). Grazed sites also tend to have lower amounts of invertebrates, the preferred food source of grasshopper sparrows (Quinn and Walgenbush 1990, Rambo and Faeth 1999, Sutter and Ritchison 2005).

Cattle also directly influence grasshopper sparrow survival through trampling of nests (Hovick *et al.* 2012). However, trampling appears to cause fewer deaths than predation (Renfrew *et al.* 2005, Hovick *et al.* 2011, 2012). Hovick *et al.* (2012) reported that only 5 of 327 (1.5%) of nests in their Iowa study failed as a result of cattle trampling. Renfrew *et al*. (2005) reported that nine of 78 (11%) grasshopper sparrow nests with eggs and four of 54 (7%) of grasshopper nests with nestlings were trampled by cattle. Number of nests destroyed due to cattle trampling may have more to do with herd behaviour than stocking rates (Renfrew *et al.* 2005).

7.5 Chestnut-collared Longspur Response to Grazing

Chestnut-collared longspurs prefer some level of disturbance, reaching their highest abundance in grassland communities that are periodically grazed and disappearing from communities that do not receive occasional disturbance (Maher 1973, Owens and Myres 1973, Dale 1983, 1984, Anstey *et al*. 1995, Davis *et al.* 1999, Dechant *et al.* 2002f, Lusk and Koper 2013). Throughout its range, chestnut-collared longspurs generally respond positively to moderate grazing in taller grasslands and negatively to heavier grazing in shorter grasslands (Bock *et al.* 1993). Kantrud and Kologiski (1983) described the chestnut-collared longspurs as showing variable responses to grazing depending on the geographic location, with heavy and moderate grazing generally preferred to light grazing in the northern portion of the Great Plains.

In Alberta, Owens and Myres (1973) observed that chestnut-collared longspurs in the Hand Hills of Alberta preferred grazed fields and avoided idle fields. Wershler *et al.* (1991) found that chestnut-collared longspurs in Alberta preferred moderately to heavily grazed mixedgrass prairie. In Grasslands National Park in Saskatchewan, Sliwinski and Koper (2015) observed that chestnut-collared longspurs increased in abundance as stocking rates increased, and were most abundant after two years in fields with the highest stocking rates. Similarly, Sliwinski (2011), also working in the park, observed greater abundances of chestnut-collared longspurs at higher stocking rates compared to lower stocking rates and ungrazed controls, with longspurs remaining relatively stable until a threshold was reached at 0.40 AUM/ha, after which abundance increased. At Matador, Saskatchewan, both Maher (1979) and Bleho *et al*. (2015) noted higher breeding chestnut-collared longspur densities in grazed fields compared to ungrazed fields. Similarly, Fairfield (1968, cited in Bleho *et al*. 2015) found that chestnut-collared longspurs in Saskatchewan were more common on over-grazed fields compared to adjacent lightly grazed fields (both mixedgrass prairie).

In the United States, Fritcher *et al.* (2004) observed higher densities of chestnut-collared longspurs in early seral stage native mixedgrass prairie in South Dakota. In fact, densities were 25 times higher in early seral stage plant communities than in late seral stage plant communities. Early seral stage communities were dominated by buffalo grass (*Buchloe dactyloides*) and western wheat grass. Kantrud (1981) observed that chestnut-collared longspurs in North Dakota preferred heavily grazed fields followed by moderately grazed fields, lightly grazed fields and hay fields, which were less preferred. Preferred grazing intensity may differ by soil type (Kantrud and Kologiski 1982). In contrast to other studies linking grazing to chestnut-collared longspur occurrence, Davis *et al.* (1999) found that chestnut-collared longspurs in Saskatchewan were relatively insensitive to grazing intensity. Similarly, livestock grazing may not be a crucial habitat variable for chestnut-collared longspurs on their non-breeding range in the semiarid grasslands of central New Mexico (Kelly *et al.* 2006).

Livestock grazing may also influence chestnut-collared longspur nesting success. Pipher (2011) observed a non-linear correlation of stocking rate on nesting success of chestnut-collared longspurs in Saskatchewan, with nesting success higher in ungrazed and heavily grazed fields. Nesting success of chestnut-collared longspurs was also negatively correlated with the number of years a field was grazed (Pipher 2011). In contrast, Lusk (2009) found no correlation between long-term grazing and nesting success of chestnut-collared longspurs in the same study area. In

southern Alberta, Koper and Schmiegelow (2007) observed that chestnut-collared longspurs had lower nesting success in ungrazed fields compared to fields that were deferred grazed.

Trampling of chestnut-collared longspur nests by livestock appears to be a low mortality risk. Of 469 nests chestnut-collared longspur nests monitored across the species' range in Canada, three (*i.e.*, less than 1%) were trampled by cattle (Bleho *et al.* 2014). Hill (1997) observed trampling-related mortality of only one of 269 (0.4%) nests near Bindloss, Alberta. Yoo (2014) observed only two of 155 (1.3%) nests were destroyed by cattle in her study area near Brooks, Alberta. Trampling effects appear to increase as cattle grazing intensity increases (Bleho *et al.* 2014).

8 GRAZING SYSTEMS AND GRASSLAND BIRD GROUP 2 HABITAT MANAGEMENT

A number of studies have examined the effects of different grazing systems on grassland birds of North America. Various grazing systems and their potential effects on grassland birds are summarized in Table III-8. It should be stressed that there is no 'one-size-fits-all' approach or 'best' system that will work in all circumstances. Each ranch and lease has their own unique circumstances and will need to be approached differently. It is recommended to apply an adaptive approach that balances ranch/lease needs with the needs of grassland birds for certain habitat types (Adams *et al.* 2004). Grazing systems should be flexible and will ideally evolve over time to minimize their negative effects on grassland birds.

Table III-8 Grazing Systems and Grassland Bird Habitat Management

Grazing System	Discussion
Continuous (Season-Long) Grazing	
Advantages:	Continuous grazing systems at light to moderate stocking rates may create suitable habitat conditions for the suite of species considered in this section if it promotes variable grazing pressure across the rangeland with a gradient of light to heavily used patches. Variably grazed patches help to create a heterogeneous habitat mosaic suitable for meeting the diverse cover requirements of a range of grassland-nesting birds. Heavier grazed patches provide foraging habitat for upland sandpipers and long-billed curlews and nesting habitat for grasshopper sparrows, while areas of less use provide suitable nesting cover for Sprague's pipits, Baird's sparrows and chestnut-collared longspurs. Continuous grazing in mixedgrass habitats in Alberta may be particularly beneficial for long-billed curlews (Prescott *et al.*1993, Prescott and Wagner 1996). Prescott *et al.* (1993) and Prescott and Wagner (1996) found that long-billed curlews preferentially used continuously grazed fields in comparison to deferred grazed or tame pastures near Brooks, Alberta. Continuous grazing, unlike deferred grazing, is capable of providing long-billed curlews with suitable foraging habitat (*i.e.*, patches of shorter, open

	vegetation) early in the season. More studies are needed though to determine productivity of long-billed curlews in continuous grazing systems in the Milk River and South Saskatchewan River watersheds. In Grasslands National Park, Ranellucci *et al*. (2012) observed higher grassland bird species richness and diversity in continuously grazed fields compared to twice-over rotationally grazed and ungrazed fields. A continuous grazing system in this mixedgrass prairie landscape led to the creation of a spatially heterogeneous landscape with a diversity of high and low use areas. Working in the same study area, Lusk (2009) observed that continuous grazing at light to moderate intensities had no significant effect on nest survival for Sprague's pipits, Baird's sparrows and chestnut-collared longspurs. Higher densities of grasshopper sparrows were found under continuous grazing and short duration, twice-over rotational grazing compared to other grazing systems (*e.g.*, rotational grazing) and ungrazed fields (Messmer 1990, Temple *et al*. 1999, Johnson and Sandercock 2010).
Disadvantages:	Continuous grazing can be detrimental to grassland bird habitat if stocking rates are high, resulting in uniform reduction in vegetation or litter cover (Dechant *et al.* 2002b,c,d). Extensive removal of vegetation cover and litter reduces the available nesting cover for upland sandpipers, Sprague's pipits, Baird's sparrows, grasshopper sparrows and chestnut-collared longspurs, and potentially eliminates suitable brood-rearing habitat for long-billed curlews. The effects of heavy, continuous grazing are partially dependent on the size of continuously grazed pastures and the heterogeneity of the surrounding grassland landscape. Another negative effect of continuous heavy grazing is that it may result in increased trampling risks to nests (Kirsh and Higgins 1976, Ailes 1980, Cochran and Anderson 1987, Bowen and Kruse 1993). Under continuous grazing, defoliation occurs during the nesting period and no areas are rested. Prescott and Wagner (1996) did not observe grasshopper sparrows in continuously grazed fields as part of the Medicine Wheel project in southern Alberta, perhaps suggesting that this type of grazing system is not favourable to this sparrow species. Messmer (1990) observed higher densities of chestnut-collared longspurs on twice-over rotationally grazed fields compared to fields that were continuously or short-duration grazed.

Rotational Grazing - Deferred Rotation	
Advantages:	Grazing that is deferred during the critical brood-rearing and nesting periods (*i.e.*, mid to late summer) is thought to promote higher nest success among most ground-nesting birds (Holechek *et al.* 1982). Deferral minimizes loss of residual nesting cover early in the season and avoids trampling disturbance to nests. Deferred grazing also has benefits for improved range health as native grasses are generally more sensitive to defoliation early in the growing season (Adams *et al.* 1991). Nesting success of upland nesting passerines in the Aspen Parkland of central Alberta was higher at the field level in fields that were grazed later in the season (*i.e.*, July) than those grazed in late May or June (Prescott *et al*. 1997). Similarly, various studies in Wyoming, North Dakota and Wisconsin have correlated reduced nest density and reduced nesting success of upland sandpipers and long-billed curlews with occurrence of livestock grazing during the nesting period (Kirsh and Higgins 1976, Ailes 1980, Cochran and Anderson 1987, Bowen and Kruse 1993). Bowen and Kruse (1993) recommended delaying grazing until after nesting is underway in- mid to late June to reduce potential impacts to upland sandpipers in mixedgrass prairie in North Dakota. In southern Alberta, Prescott and Wagner (1996) found that upland sandpipers were only present in deferred grazed pastures.
Disadvantages:	The benefits of deferred grazing are reduced if heavy use occurs subsequent to deferral, thereby diminishing brood-rearing cover and reducing residual vegetation for the next nesting season (Holechek *et al.* 1982). Another possible negative effect of deferred grazing is that it may diminish the amount of suitable foraging habitat that is available to long-billed curlews early in the season. For example, in southern Alberta no long-billed curlews were found on plots where grazing was deferred until after mid-June when pairs had already established territories (Prescott and Wagner 1996). The majority of long-billed curlews were either on early season (April to mid-June) grazed or continuously grazed plots (Prescott and Wagner 1996) Prescott and Wagner (1996) did not observe grasshopper sparrows in native prairie fields grazed after July 15th as part of the Medicine Wheel project, perhaps suggesting that this type of grazing system is not favourable to this sparrow species. In Kansas tallgrass prairie, Johnson and Sandercock (2010) observed similar densities of grasshopper sparrows in winter-grazed and year-long continuously grazed fields.

Complementary Grazing	
Advantages:	Complementary grazing, a form of rotational grazing, also promotes undisturbed residual nesting cover and limits disturbance to birds during nesting and brood-rearing. Deferred grazing of native prairie early in the season, as mentioned, can improve the health, productivity and sustainability of fescue and mixed grasslands. Based on the results of a six year study, Prescott and Wagner (1996) concluded that complementary grazing in combination with rotational grazing improved range condition, grass yields and litter reserves in mixedgrass prairie in southern Alberta. These improvements were noted to provide a mosaic of habitats suitable for a wide spectrum of upland nesting birds, including long-billed curlews, upland sandpipers, Sprague's pipits, Baird's sparrows, grasshopper sparrows and chestnut-collared longspurs. Prescott and Wagner (1996) observed grasshopper sparrows strictly in crested wheat grass tame pastures that were grazed from late April to mid-June and not in continuously grazed or deferred grazed native prairie fields.
Disadvantages:	The benefits of complementary grazing depend on whether existing cropland or hayland can be converted to tame pasture. If tame pasture is created by converting native prairie habitat, fewer benefits may be realized. Although long-billed curlews, upland sandpipers, Baird's sparrows, grasshopper sparrows and chestnut-collared longspurs have been found to utilize tame pastures for either foraging, nesting or brood- rearing, Sprague's pipits are significantly more abundant in native prairie habitats (Robbins and Dale 1999).
Rotational Grazing – Rest Rotation	
Advantages:	Rotational grazing is often recommended as an effective grassland bird habitat management tool (Sedivec *et al.* 1991, Bowen and Kruse 1993, Prescott and Wagner 1996, Stanley *et al.* 1999). Rotational grazing systems allow fields to be rested in a sequential fashion for either an entire season or for part of a season (Holechek *et al.* 2003). This provides a mechanism to promote vegetation heterogeneity at a field level and allows areas to be free from livestock disturbance during the critical nesting and brood-rearing periods. Rotational grazing systems, due to the requirement of additional fencing, have the added benefit of creating numerous perch sites for upland sandpipers (Houston and Bowen 2001). Rotational grazing systems have been promoted for long-billed curlews in Idaho (Bicak *et al.* 1982), for upland sandpipers and Baird's sparrows in North Dakota (Messmer 1990, Bowen and Kruse 1993, Sedivec 1994) and for Sprague's pipits in mixedgrass and fescue prairie in southern Alberta (Mahon 1995, Prescott and Wagner 1996). Messmer (1990) observed higher densities of grasshopper sparrows under continuous grazing, twice-over rotational and short-duration grazing compared to other grazing systems and idle lands in North Dakota. This same author also observed higher densities of chestnut-

	collared longspurs on twice-over rotationally grazed fields compared to fields that were continuously or short-duration grazed.
Disadvantages:	Areas that are rested for periods of one year or greater may not offer suitable foraging habitat for long-billed curlews or upland sandpipers (Bicak *et al.* 1982). Grazing intensities greater than 50% utilization may diminish the benefits of rotational grazing, leading to deteriorated range health, removal of residual nesting and brood-rearing cover and uniform vegetation structure (Kobriger 1980, Holechek *et al.* 1982). In Wisconsin, Temple *et al.* (1999) observed greater abundances of grasshopper sparrows in continuously grazed fields compared to ungrazed fields or rotationally grazed fields. In Grasslands National Park, Ranellucci *et al*. (2012) observed higher grassland bird species richness and diversity in continuously grazed fields compared to twice-over rotational grazing and ungrazed fields.
Rotational Grazing – High Intensity Grazing	
Advantages:	Intensive grazing may be appropriate for creating suitable habitat conditions for long-billed curlews and upland sandpipers in mesic grasslands with higher productivity or in tame pastures (Skinner 1975, Kantrud and Kologiski 1982, Bolster 1990). In North Dakota, Messmer (1990) found the highest breeding densities of upland sandpipers under a short-duration grazing system in mixedgrass prairie in south-central North Dakota. Both long-billed curlews and upland sandpipers use heavier grazed areas for foraging (Dechant *et al.* 2002a,b).
Disadvantages:	Intensive grazing may be particularly harmful to Baird's sparrows and Sprague's pipits. Both species have consistently been shown to decline under heavy grazing pressure in mixedgrass and fescue prairie in Alberta and Saskatchewan (Owens and Myres 1973, Maher 1979, Dale 1984, Anstey *et al.* 1995). Heavy grazing may also reduce or eliminate suitable brood-rearing and nesting habitat for long-billed curlews and upland sandpipers, respectively, particularly in arid mixedgrass regions (Dechant *et al.* 2002a,b). Kantrud (1981) noted an overall trend of reduced bird species diversity with increased grazing intensity in North Dakota. Ground- nesting birds are also potentially vulnerable to increased rates of nest abandonment or trampling with increasing stocking rates (Hill 1998). The uniform use created by intensive grazing is unlikely to benefit grasshopper sparrows or chestnut-collared longspurs due to their preference for habitat heterogeneity (Davis 2005, Hovick *et al.* 2012).

9 BENEFICIAL MANAGEMENT PRACTICE RECOMMENDATIONS

The following general land-use and grazing recommendations offer a variety of means to protect or enhance grassland bird habitat within the Milk River and South Saskatchewan River watersheds and throughout the Grassland Natural Region of Alberta. Further research is required to improve our understanding of these species and their habitat requirements as well as their response to various grazing management practices (see Section 10).

9.1 General Recommendations

Breeding Habitat Protection and Enhancement

- Protect remaining large, contiguous tracts of native prairie from cultivation (Owens and Myres 1973, Goosen *et al.* 1993, Hill 1998, Saunders 2001, Dechant *et al.* 2002a,b,c,d,). Croplands do not provide suitable habitat for Sprague's pipits and are used less and considered less productive habitats for Baird's Sparrows, long-billed curlews, upland sandpipers, grasshopper sparrows and chestnut-collared longspurs (Dechant *et al.* 2002a,b,c,d,e,f). Small patches of native prairie with minimal edge habitat also play a vital role in the conservation of grassland birds (Davis 2004).
- Encourage no-till or minimum tillage and organic farming practices over conventional tillage (Lokemoen and Beiser 1997, Houston and Bowen 2001, Dechant *et al.* 2002a,b,c,d, Martin and Forsyth 2003). These practices help to reduce or avoid disturbance during the nesting season. Martin and Forsyth (2003) found an overall higher abundance of songbirds in minimum versus conventional tilled habitats in southern Alberta. In cropland areas, a mix of minimum tillage fields in summer fallow, winter wheat (*Triticum aestivum*) and spring cereals will likely benefit the greatest number of grassland bird species (Martin and Forsyth 2003).
- Delay haying or harvesting crops until after the peak grassland bird nesting period to reduce potential losses of nests or young broods (Dale *et al.* 1997, Dechant *et al.* 2002a,b,c,d). For example, delay haying tame grasses/legumes until after July 15th and avoid harvesting cereal crops until August.
- Where practical, reclaim or restore cropland, hayland or tame pasture to native grassland (Dale *et al.* 1997, Dechant *et al.* 2002c). Give priority to restoring areas adjacent to high quality sites to counter the effects of habitat fragmentation (Dale *et al.* 1997). Develop priority maps for land reclamation/restoration.
- Control encroachment of broad-leaved exotic graminoids such as smooth brome and timothy in native prairie habitats. Sprague's pipits and Baird's sparrows prefer habitats with narrow-leaf grasses and have been negatively associated with broad-leaved graminoids (Wilson and Belcher 1989, Dale 1992, Anstey *et al.* 1995).
- Promote habitat heterogeneity of grasslands by implementing controlled burns, grazing or haying treatments, allowing for sufficient rest periods (Dechant *et al.* 2002a,b,c,d). Provide a mosaic of vegetation heights and densities and variable grass to forb cover ratios to meet the diverse habitat requirements of long-billed curlews and upland sandpipers over the course of the breeding season. Allow some areas to be undisturbed during the nesting season.

- Maintain sufficient patch sizes of open grassland (see Area Requirements, below) with minimal (*i.e.*, less than 20%) shrub or woody species encroachment (Dechant *et al.* 2002a,b,c,d). Long-billed curlews, upland sandpipers, Sprague's pipits, Baird's sparrows, grasshopper sparrows and chestnut-collared longspurs all tend to avoid grasslands with dense shrub cover.
- Use Protective Notations under the *Public Lands Act* to protect key grassland bird habitats.
- Work with landowners and conservation groups to conserve habitat for grassland birds on private land in southern Alberta.

Area Requirements

- Protect large, contiguous blocks of native prairie habitat. Habitat fragmentation can cause patch-size, edge and isolation effects that can negatively affect use or reproductive success of breeding birds (Johnson and Igl 2001, Davis 2004). Several studies have shown that nest predation and nest parasitism rates are higher in edge habitats (Johnson and Temple 1990). Moreover, studies have found that grassland breeding bird species richness increases significantly with increasing patch size (Herkert 1994, Vickery *et al.* 1994).
- Where possible, manage and maintain habitat parcels of at least 100 ha (and preferably 200 ha) in size to reduce edge effects, decrease nest predation and provide sufficient habitat heterogeneity (Herkert 1994, Vickery *et al.* 1994, Houston and Bowen 2001). Blocks of habitat should be situated within 1 km of each other, and should be contiguous with grassy habitats such as tame pastures or hayfields (Herkert 1994, Vickery *et al.* 1994, Walk and Warner 1999).
- Consider patch shape and core area dimensions (Helzer and Jelinski 1999). Maintain or create grassland habitats with high interior area and minimized edge (*i.e.*, low perimeter to area ratio) (Helzer and Jelinski 1999). For example, irregularly-shaped patches have higher perimeter to area ratios than circular or regular shaped patches (Helzer and Jelinski 1999). Helzer and Jelinski (1999) found that patch area and low perimeter-area ratios were positive predictors of grassland bird species richness.

Prescribed Burning and Mowing

- In the absence of grazing, prescribed burning or mowing provide alternative management tools to control woody species encroachment, reduce dense litter build-up and create a mosaic of grasslands of different heights (Hull 2002, Dechant et al. 2002a,b,c,d,f). These factors are beneficial for creating suitable habitat for long-billed curlews, upland sandpipers, Sprague's pipits, Baird's sparrows, grasshopper sparrows and chestnut-collared longspurs (Dechant et al. 2002a,b,c,d,e,f). By improving structural heterogeneity, this form of management also promotes suitable habitat for a diversity of other grassland birds. It can also be used to improve forage quality and availability for livestock in highly productive, mesic grasslands. Burning or mowing treatments may not be appropriate or desirable in arid conditions where accumulation of litter or tall, dense residual vegetation is not a concern (Madden *et al.* 1999, Dechant *et al.* 2002d).

- The timing, spatial extent and distribution, and frequency of burns or mows are key considerations to minimize disturbance and avoid possible losses of eggs or young. Frequent, large scale burns can also decrease forb species diversity and reduce grassland nitrogen levels (Arenz and Joern 1996).
- Burning and mowing should be deferred until after the peak nesting season (*i.e.*, from May 1 to the end of July) (Mahon 1995, Patterson and Best 1996, Dale *et al.* 1997, Dechant *et al.* 2002a,b,c,d). Where possible, delay mowing or burning into mid-August to minimize possible nest or brood losses, particularly during years of delayed nesting such as years of inclement spring weather (Dale *et al.* 1997, Dechant *et al.* 2002d).
- Mow grasslands or cut hay on a rotational schedule whereby fields are mowed every second year at a delayed date (Dale *et al.* 1997). Alternatively, divide large hay fields in half, with each half mowed in alternate years (Dale *et al.* 1997). This will help minimize potential nest losses and provide adjacent protective cover for newly fledged young in mowed areas (Dale *et al.* 1997). Hay fields should not be idled for more than one year to remain suitable as breeding sites for Sprague's pipits or Baird's sparrows (Dale *et al.* 1997).
- Prescribed burning should also be conducted on a rotational basis, with recently burned patches adjacent to unburned or less recently burned areas. This will help to create patches with a variety of successional stages and vegetation heights (Renken and Dinsmore 1987, Johnson 1997, Madden *et al.* 1999, Dechant *et al.* 2002d). Herkert (1994) recommends burning 20% to 30% of grassland fragments less than 80 ha in size to ensure that sufficient amounts of unburned area are left to provide nesting or brood-rearing cover.
- Burn intervals should correspond with the approximate historic fire return interval for the region (Madden *et al.* 1999). The estimated historic fire return interval is 5 to 10 years for fescue prairie (Arno 1980, Wright and Bailey 1982), 6 years for northern mixedgrass prairie and up to 25 years for dry, western mixedgrass prairie (Madden *et al.* 1999).
- Prescribed burns should be conducted either early in the season (*i.e.*, March to early April) or late in the season (*i.e.*, October to November) (Kirsh and Higgins 1976, Herkert 1994).

Pest Control

- Reduce or avoid the use of organochlorine pesticides, including carbamates, carbofurans and organophosphates (De Smet 1992a, COSEWIC 2002a). The use of organochlorine pesticides has caused documented mortalities and other sublethal effects in long-billed curlews (Blus *et al*. 1985). More research is needed to determine the potential detrimental effects of pesticide use on other grassland bird species (Prescott and Davis 1999, Houston and Bowen 2001, Green *et al.* 2002). Pesticide use may adversely affect grassland bird breeding success due to direct chemical ingestion or reduction in invertebrate prey abundance (De Smet 1992a, Prescott and Davis 1999).
- Encourage organic farming practices and promote grassland birds as natural pest control agents.

Anthropogenic Disturbance Mitigation

- Abide by AEP recommended set-back distances and timing restrictions for long-billed curlews, upland sandpipers and Sprague's pipits (GoA 2011). Nests of these three species, including surrounding habitat, have set-back distances of 100 m from April 1st to July 15th for low, medium and high disturbance activities, including oil and gas developments and agricultural and recreational activities.
- Conduct pre-development wildlife surveys to locate grassland birds in areas with suitable habitat. Ensure wildlife survey data is entered into the Fish and Wildlife Management Information System (FWMIS) database maintained by AEP. Nest searches, at least for chestnut-collared longspurs, are best conducted early in the morning and late in the evening as these are the times of day when females are most likely to be flushed while incubating nests (Kirkham and Davis 2013).
- Enforce provisions under the *Wildlife Act* and *Migratory Birds Convention Act* protecting grassland birds and their nests and eggs from disturbance.
- Reduce human disturbance in breeding areas during the nesting period to avoid disturbing nesting birds and young broods (Dechant *et al.* 2002a). Long-billed curlews are considered particularly sensitive to human disturbance during the nesting and brood rearing periods (Dugger and Dugger 2002). Excessive off-road vehicle use, for example, can cause nest abandonment and disrupt brooding or shading of chicks by long-billed curlew adults (Dugger and Dugger 2002).
- Minimize road development associated with oil and gas and other industrial developments in prime native prairie habitats to reduce habitat loss and fragmentation (Redmond and Jenni 1986, Hill 1998). Preliminary research suggests that roads, as opposed to the wells themselves, may have a negative effect on grassland bird nest survival through creation of edge habitat and travel corridors for predators (Yoo 2014). Early evidence also suggests that high well densities may be particularly harmful to chestnut-collared longspur reproductive success, and longspur clutch sizes and fledgling numbers per nest may be lower near gas well pads (Yoo 2014). Grasshopper sparrows have also been shown to decrease in abundance in close proximity to gas wells (Bogard and Davis 2014). In contrast, Sprague's pipit and Baird's sparrow abundance may not be affected by gas well density or proximity (Bogard and Davis 2014).
- Avoid the construction of new dams in southern Alberta. Dams lead to flooding of upland habitats as the reservoirs fill up. This could lead to loss of native prairie habitat required by grassland songbirds for nesting and foraging (Environment Canada 2012).

Population Monitoring and Data Needs

- Develop and implement standardized, long-term population monitoring programs for grassland songbirds in southern Alberta. Ensure all observations are submitted for inclusion in the provincial Fish and Wildlife Management Information System (FWMIS). Consider reporting 'zero data' as well.
- Prepare provincial species at risk status reports for the upland sandpiper, Baird's sparrow, grasshopper sparrow and chestnut-collared longspur. Update the status reports for the long-billed curlew and Sprague's pipit.

- Continue to refine provincial population estimates for the long-billed curlew, uplands sandpiper, Baird's sparrow, Sprague's pipit, grasshopper sparrow and chestnut-collared longspur.

9.2 Grazing Recommendations

Livestock grazing can be used as a tool to manage for grasslands with diverse vegetation structure and species composition, suitable to a broad range of grassland birds. Appropriate grazing systems should be developed site-specifically for participating ranches as part of the MULTISAR conservation program in southern Alberta.

Grazing systems that will benefit long-billed curlews, upland sandpipers, Sprague's pipits, Baird's sparrows, grasshopper sparrows and chestnut-collared longspurs should aim to:

- Create a mosaic of variably grazed patches across the landscape, with a gradation of heavy, moderate and light use.
- Leave at least one field or a portion of a field undisturbed during the peak nesting period (*i.e.*, May to mid-July).
- Maintain shrub canopy cover at less than 20%.
- Stimulate vegetation diversity with moderate grass to forb ratios, a mixture of tall, medium and short canopy grass species and scattered tall forbs (Mahon 1995).

In order to meet these objectives, the following grazing recommendations apply:

- Avoid intensive, high stocking rate grazing systems and discourage uniform utilization of fields (Dechant *et al.* 2002a,b,c,d). Uniform heavy use diminishes nesting or brood-rearing cover and may result in greater disturbance to nesting birds.
- Use appropriate stocking rates and proper utilization rates to ensure sufficient carry-over and maintain plant vigour and range condition (Hull 2002, Dechant *et al.* 2002a,b,c,d). In general, apply light to moderate stocking rates and utilization rates of 25% to 50% for native grassland plant communities in southern Alberta (Adams *et al.* 2005, 2013a,b). Adjust stocking rates in accordance with range health, natural subregion, ecological site and management objectives. If the management goal is to benefit the greatest number of grassland songbird species, then applying a range of stocking rates regionally will be most beneficial (Fritcher *et al.* 2004, Sliwinski and Koper 2015).
- Make seasonal adjustments to stocking rates based on fluctuations in precipitation, time of use and range and riparian health.
- Strategically place salt and water sources to encourage cattle distribution across the landscape (Adams *et al.* 1986).
- Develop a salt placement plan for each field to selectively manage for areas of higher use. Chestnut-collared longspurs, for example, prefer moderate to heavy grazing intensities (Wershler *et al.* 1991, Dechant *et al.* 2002f).
- Defer the use of native prairie early in the spring using either deferred rotation or complementary grazing systems. Deferred early-season use of native prairie can benefit range health and minimize disturbance to nesting birds (Prescott and Wagner 1996).

Complementary grazing systems are best-suited to operations where tame pastures exist or can be created without loss of native prairie (Prescott and Wagner 1996).

- Promote the use of rotational grazing systems that allow fields to be rested for either an entire season, or for part of the season (Messmer 1990, Sedivec 1991, Bowen and Kruse 1993, Mahon 1995, Prescott and Wagner 1996, Stanley *et al.* 1999). Rotational grazing systems can be used to create variable vegetation structure at the field level and provide a means to delay disturbance in at least one field during the peak nesting season.
- Continuous grazing at light to moderate stocking rates can also be conducive to maintaining heterogeneous habitat that benefits the greatest number of grassland songbirds (Ranellucci *et al*. 2012)
- Where possible, winter graze fescue grasslands at recommended stocking rates according to ecological site and range health status (Mahon 1995, Hull 2002). Mahon (1995) recommended November and December grazing of fescue grasslands in the Milk River Ridge area as a suitable strategy for maintaining habitat for Baird's sparrows. Winter grazing avoids use during the nesting period, helps to reduce dense accumulations of litter and residual vegetation and may also be more effective for reducing woody cover (Medin 1986, Mahon 1995). Winter grazing is also a recommended strategy to maintain the health and productivity of rough fescue grasslands that are better adapted to later season use (Adams *et al.* 1994).

10 RESEARCH RECOMMENDATIONS

On-going monitoring of grassland bird populations in the Milk River and South Saskatchewan River watersheds is necessary to continue to assess population trends and bird response to land-use practices. The Grassland Bird Monitoring Pilot Project that was initiated in 2002 may provide a means to more effectively monitor and track trends of grassland birds in the Canadian prairies (Dale *et al.* 2002). This project incorporated information on habitat in conjunction with bird surveys. Long-term monitoring of this kind should be coupled with on-going applied research to better understand various aspects of grassland bird ecology and response to land-use activities.

Key research needs for long-billed curlew, upland sandpiper, Sprague's pipit, Baird's sparrow, grasshopper sparrow and chestnut-collared longspur populations in the Milk River and South Saskatchewan River watersheds are listed below:

- Continue to assess grassland bird response to habitat fragmentation and determine minimum area requirements (Hill 1998).
- Examine the effects of pesticide and herbicide use on survival, nesting success and prey availability.
- Conduct additional basic research to better understand the nesting ecology and nesting habitat characteristics of long-billed curlews, upland sandpipers, Sprague's pipits, Baird's sparrows, grasshopper sparrows and chestnut-collared longspurs in the Grassland Natural Region of Alberta (Sutter 1997, Green *et al.* 2002, NRC 2006).
- Collect more information on seasonal survival rates of chicks, subadults and adults to quantify sources of mortality (Dugger and Dugger 2002).

- Compare relative use and nesting and fledgling success in dry mixedgrass, mixedgrass and fescue prairie managed under different grazing systems and intensities of use.
- Determine long-term trends of grassland bird populations in response to changes in range health.
- Assess and compare the effects of different intensities and frequencies of burning, grazing and mowing on nest site selection and productivity (De Smet and Conrad 1991).
- Compare grassland bird use and productivity in tame pastures versus native prairie habitats in the Milk River and South Saskatchewan River watersheds. Determine if anthropogenic habitats such as tame pastures act as ecological sources or sinks (Environment Canada 2012, 2016).
- Compare rates of nest predation and brown-headed cowbird parasitism in native versus non-native habitats and in varying patch sizes in the Milk River and South Saskatchewan River watersheds.
- Determine the effects of habitat patch size on grassland bird reproductive success.
- Determine the extent of grassland bird use of restored/reclaimed grasslands and what factors are critical to creating viable new habitat (Environment Canada 2012, 2016).
- Conduct further research into the effects of oil and gas activity on grassland bird density, survival and productivity.
- Determine the effects of wind energy development on the density, survival and productivity of grassland songbirds (Environment Canada 2016).
- Determine the effects of climate change on the density, survival and productivity of grassland songbirds (Environment Canada 2016).
- Develop Resource Selection Function (RSF) models for the six grassland bird species discussed in this section in order to identify potential critical habitat. Consider aggregating individual grassland bird models to prioritize the landscape in southern Alberta according to multi-species conservation values. Conduct field verification of the grassland bird RSF models to determine their accuracy.
- Determine thresholds for critical habitat biophysical attributes for grassland songbirds, including percent shrub cover, exotic species invasion, degree of anthropogenic disturbance and topographic relief (Environment Canada 2016).

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G. BREWER'S SPARROW

1 INTRODUCTION

The purpose of this report is to summarize and compare the ecology and habitat requirements of the Brewer's sparrow (*Spizella breweri*) in southern Alberta. The Brewer's sparrow is different from other grassland songbirds discussed in this report due to its general preference for shrubland (as opposed to grassland) for nesting (Rich 1980, Reynolds 1981, Wiens and Rotenberry 1981). Based on information on the species' ecology and habitat preferences, livestock grazing interactions and the potential effects of various grazing systems on Brewer's sparrows and its habitat are discussed. This discussion is followed by a summary of recommended beneficial management practices (BMPs) to enhance habitat for this species in the Milk River and South Saskatchewan River watersheds in Alberta and throughout their range in the Grassland Natural Region of Alberta (Natural Regions Committee [NRC] 2006). Lastly, a brief summary of additional research needs is presented.

1.1 Background

The Brewer's sparrow is a small, slender sparrow with a brown, finely black-streaked crown, a white eye ring and pale brown ear patches (Semenchuk 1992). It is similar to the more common clay-colored sparrow (*Spizella pallida*), but it lacks the white crown stripe, grey hind neck and distinct facial markings of that species (Semenchuk 1992). The Brewer's sparrow occurs primarily in western North America from Alaska south to Mexico (Rotenberry *et al.* 1999). There are two subspecies of Brewer's sparrow in Alberta: *Spizella breweri breweri*, which occurs primarily in the Grassland Natural Region; and *S. b. taverneri*, which occurs primarily in the Rocky Mountain Natural Region (Semenchuk 1992, NRC 2006, Federation of Alberta Naturalists 2007). The two subspecies are separated by geography and habitat preference as well as by biological and behavioural characteristics (Semenchuk 1992). *S. b. breweri* occurs in the southern portion of the province as far north as Buffalo Lake, while *S. b. taverneri* occurs in the mountains from Jasper National Park in the north to Waterton Lakes National Park in the south (Semenchuk 1992, Federation of Alberta Naturalists 2007).

The Brewer's sparrow is currently listed as 'Sensitive' under the Alberta General Status listing (Government of Alberta [GoA] 2016). The species is listed as a 'non-game animal' under the provincial *Wildlife Act*. Brewer's sparrows and their eggs and nests are also protected from harm under the federal *Migratory Birds Convention Act*. The Brewer's sparrow has not been assessed nationally by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) and the species is not included under Schedule 1 of the *Species at Risk Act* (*SARA*) (COSEWIC 2016). Internationally, the Brewer's sparrow is listed as a species of 'Least Concern' by the International Union for Conservation of Nature and Natural Resources (IUCN) (BirdLife International 2012).

The Brewer's sparrow has experienced steep population declines throughout its range in North America since the mid-1990s (Rotenberry *et al.* 1999, GoA 2016). In Alberta, the species relies on the availability of silver sagebrush (*Artemisia cana*) plant communities. Conversion of native prairie with silver sagebrush to cropland is thought to be the reason for the decline of the

Brewer`s sparrow in Alberta (GoA 2016). Formal status reports have yet to be prepared for the Brewer's sparrow by either AEP or COSEWIC.

1.2 Ecology

Brewer's sparrows arrive in southern Alberta in early to mid-May and in the mountains in late May (Nordin *et al.* 1988, Semenchuk 1992). Both subspecies migrate south in August and September (Semenchuk 1992). Pairs form within a few days of males and females reaching their breeding grounds (Nordin *et al.* 1988). Males undertake courtship feeding to secure pair-bonds with females (Nordin *et al.* 1988). Nests are cup-shaped, made of grass, roots and weed stems, and are placed low to the ground in shrubs (Semenchuk 1992). Clutches in southeastern Alberta are initiated in late spring/early summer (between May 26th and June 27th; Biermann *et al.* 1987). Clutches contain three to five eggs, each of which is greenish-blue with reddish-brown dots (Semenchuk 1992). Mahony *et al.* (2006) reported average clutch sizes in British Columbia of 3.76 ± 0.02 eggs, whereas Reynolds (1981) reported average clutch sizes of 3.4 ± 0.5 eggs in Idaho. Females incubate the eggs for 11 to 13 days, and young birds fledge after eight or nine days (Semenchuk 1992). Average brood size of successful nests in British Columbia was 3.27 ± 0.04 young (Mahony *et al.* 2006). Reynolds (1981) reported an average of 0.5 fledglings per nest in Idaho, with only 14% of nests fledging at least one young.

Petersen and Best (1987) reported daily nest survival ranging from 0.93 to 0.99 over five years in Idaho. Nest predation has been identified as the primary factor responsible for reproductive failure in grassland birds in general (Best *et al.* 1997, Koford 1999, Jones *et al.* 2010) and in Brewer's sparrow in particular (Mahony *et al.* 2006, Ruehmann *et al.* 2011). Brewer's sparrows often re-nest if the first nest is abandoned and often produce multiple broods per season (Rotenberry *et al.* 1999, Mahony *et al.* 2006). Overall, seasonal fecundity in Brewer's sparrows depends on climate, rates of nest predation and rates and success of re-nesting (Mahony *et al.* 2006).

Male Brewer's sparrows often appear to show fidelity to breeding territories, returning to the same locations year after year (Rotenberry *et al.* 1999). Although little is known about female site fidelity, males and females often form pair-bonds that persist from year to year (Walker 2004). Brewer's sparrows are not known to show fidelity to natal sites (Rotenberry *et al.* 1999).

1.2.1 Diet

Brewer's sparrows forage in shrubs, gleaning insect prey off of foliage (Wiens *et al.* 1987). Nestling Brewer's sparrows in big sagebrush (*Artemisia tridentata*) habitat in Idaho were fed exclusively arthropods by their parents (Peterson and Best 1986). Orders of insects consumed by Brewer's sparrow nestlings included Hemiptera/Homoptera (true bugs) (29%), Lepidoptera (butterflies and moths) (24%), Aranae (spiders) (19%) and others, such as Hymenoptera (ants, bees and wasps), Psocoptera (booklice, barklice and barkflies), Diptera (true flies), Coleoptera (beetles) and Orthoptera (grasshoppers and crickets), each of which constituted less than 10% of the diet.

Adult Brewer's sparrow diets in Montana in summer (*i.e.*, June and July) were comprised primarily of insects of the orders Coleoptera, Hemiptera, Orthoptera, Aranae, Hymenoptera and Lepidoptera (Best 1972). Seeds constituted between 8% and 17% of the summer diet, including seeds from species such as green needle grass (*Nassella viridula*), needle-and-thread (*Hesperostipa comata*), muhly (*Muhlenbergia* spp.) and Sandberg bluegrass (*Poa secunda*) (Best 1972).

In Alberta, Brewer's sparrows feed primarily on insects, spiders and seeds during the breeding season and almost exclusively on seeds during the winter (Semenchuk 1992).

1.2.2 Predators

Loggerhead shrikes prey on both adult and nestling Brewer's sparrows in Idaho, and may be an important determinant of sparrow nesting success, despite birds comprising only a small portion of the diets of shrikes (Reynolds 1979). Other potential predators of Brewer`s sparrows in Idaho include gopher snakes (*Pituophis melanoleucus*), prairie rattlesnakes (*Crotalus viridis*), least chipmunks (*Neotamias amoenus*) and long-tailed weasels (*Mustela frenata*). In Wyoming, known predators of Brewer's sparrow nests include deer mice (*Peromyscus maniculatus*), least chipmunks, wandering garter snakes (*Thamnophis elegans*) and elk (*Cervus elaphus*) (Ruehmann *et al.* 2011). Other potential predators noted by Ruehmann *et al.* (2011) for Wisconsin include the sage thrasher (*Oreoscoptes montanus*), black-billed magpie (*Pica hudsonia*), loggerhead shrike, brown-headed cowbird, coyote (*Canis latrans*) and small rodents. Wiens and Rotenberry (1998) noted that American kestrels (*Falco sparverius*), prairie falcons (*Falco mexicanus*) and loggerhead shrikes are known predators of adult Brewer's sparrows in Oregon.

Brewer's sparrow nests are often parasitized by brown-headed cowbirds (Rich 1978, Reynolds 1981, Mahony *et al.* 2006). Biermann *et al.* (1987) reported that 13 of 25 (52%) of the Brewer's sparrow nests they observed in southeastern Alberta were parasitized by brown-headed cowbirds, with nine of the 13 nests subsequently abandoned. Mahony *et al.* (2006) reported that 3.5% of nests in their British Columbia study area were parasitized by brown-headed cowbirds.

1.3 Habitat Requirements

1.3.1 General

Brewer's sparrows are species of the North American shrub-steppe, mixedgrass prairie and semi-desert shrub-steppe ecosystems, preferring flat, open land with suitable, homogeneous shrub cover, primarily sagebrush (*Artemisia* spp.) (Rotenberry and Wiens 1980, Wiens and Rotenberry 1981, Knopf *et al.* 1990, Rotenberry *et al.* 1999). They are sometimes referred to as 'sagebrush-obligate' species due to their general preference for sagebrush habitat (Rotenberry *et al.* 1999). In Alberta, *S. b. breweri* inhabits semi-arid plains with short grasses, cacti and low shrubland dominated by silver sagebrush and silverberry (*Elaeagnus commutata*) (Semenchuk 1992). *S. b. taverneri* inhabits mountain meadows at timberline, particularly those meadows with willows (*Salix* spp.) and birch (*Betula* spp.), or subalpine fir (*Abies bifolia*) krummholz.

It is not known if Brewer's sparrows are selecting for sagebrush in particular or shrubs in general (Rotenberry and Wiens 1998). Several authors found no correlation between sagebrush and

Brewer's sparrow occurrence (Wiens and Rotenberry 1981, Knick and Rotenberry 1995). For example, Wiens and Rotenberry (1981) found that Brewer's sparrows were not positively correlated with big sagebrush cover in the Great Basin of the United States. However, the species was negatively associated with both spiny hopsage (*Grayia spinosa*) and bud sagebrush (*Picrothamnus desertorum*) (Wiens and Rotenberry 1981), neither of which occurs in Alberta. Brewer's sparrows fluctuated in abundance following herbicide treatment of shrubs in a sagebrush community in Oregon, first decreasing but then increasing in numbers (Wiens and Rotenberry 1985).

In contrast, Knopf *et al.* (1990) found that, out of 14 species of shrubs examined, Brewer's sparrows were most strongly associated with sagebrush throughout an east-west gradient across the intermountain shrub-steppe in the United States. Best (1972) observed substantial declines in Brewer's sparrows following total sagebrush kill with herbicide in Montana. Kerley and Anderson (1995) observed higher numbers of Brewer's sparrows in Wyoming in untreated Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*) habitat compared to sagebrush habitat that was treated with herbicide or that had a prescribed burn. Untreated sites had significantly higher sagebrush cover, burned sites had higher grass cover and herbicide-treated sites had higher cover of bare ground and litter. Schroeder and Sturges (1975) found that Brewer's sparrow use of big sagebrush stands sprayed with herbicide dropped by 67% and 99% one and two years after treatment, respectively, compared to unsprayed stands, and there was no evidence of nesting in the sprayed stands.

A Resource Selection Function (RSF) model developed for Brewer's sparrow habitat use in the shrub-steppe of southwestern Idaho included two variables: shrub cover and shrub patch size (Knick and Rotenberry 1995). Relative probability of Brewer's sparrow occupancy increased with increased shrub cover and increased shrub patch size. Discriminant analysis of Brewer's sparrow habitat preferences by the same authors revealed that the species prefers higher big sagebrush cover, lower levels of disturbance and spatially similar sites (Knick and Rotenberry 1995). Knick and Rotenberry (1999) were able to predict Brewer's sparrow occurrence by mapping the distribution of large, stable and intact patches of big sagebrush in Idaho. Wiens and Rotenberry (1981) found that Brewer's sparrows in the Great Basin of the northwestern United States were negatively correlated with rock cover (rocky outcrops) and shrub species diversity.

Williams *et al.* (2011) examined habitat use by grassland birds, including Brewer's sparrows, in Colorado during the breeding season. In their study area, Brewer's sparrows were more common in Saline Lowland and Loamy ecological sites than the Sandy-Skeletal ecological site. Brewer's sparrows favoured tall, open-canopied greasewood (*Sarcobatus vermiculatus*), Wyoming big sagebrush, and spiny hopsage communities, all of which were common on Saline Lowland, Loamy and Sandy ecological sites. A regression model developed by Williams *et al.* (2011) contained three variables: plant basal gap; shrubs greater than 100 cm tall/m^2; and plant canopy height. Brewer's sparrows in Colorado favoured open plant communities with scattered tall shrubs (40 cm to 100 cm tall).

Although Brewer's sparrow may nest in tame pasture, their presence in that habitat type depends on the availability of shrubs. Tame pasture is less preferred than native prairie, but more preferred than cropland (McAdoo *et al.* 1989). For example, McAdoo *et al.* (1989) found that

Brewer's sparrows in Nevada were 1.8 times more abundant, on average, in undisturbed shrub-steppe sagebrush communities with 18% to 21% shrub cover compared to 20 to 30 year old crested wheat grass (*Agropyron cristatum*) tame pastures re-colonized with 1% to 12% shrub cover. Naturalizing crested wheat grass pastures also had significantly higher abundances of Brewer's sparrows than recently-seeded crested wheat grass monocultures (McAdoo *et al.* 1989).

Wiens *et al.* (1987) observed little change in habitat use by Brewer's sparrows throughout the course of the breeding cycle.

1.3.2 Nesting Habitat

Brewer's sparrows are shrub-nesting birds (Rich 1980, Reynolds 1981, Wiens and Rotenberry 1981, Semenchuk 1992). Average cover of shrubs in the vicinity of Brewer's sparrow nests reported in the literature ranges from 26% to 42% (not including burn treatments) (summarized by Walker 2004). Peterson and Best (1985) observed that Brewer's sparrow nests were situated in plant communities with higher big sagebrush cover, taller sagebrush shrubs, lower herbaceous cover and lower bare ground cover than in surrounding areas. Suitable nesting habitat includes sagebrush-dominated shrublands with an average shrub height of 0.5 m to 1.5 m and with greater than 10% average shrub cover (Dobler *et al.* 1996, Sarell and McGuiness 1996, Rotenberry *et al.* 1999). In shrub-steppe habitat, Brewer's sparrow abundance decreases as average shrub cover decreases below 10% to 13%, and sparrows may disappear entirely when average shrub cover decreases below 3% to 8% (Dobler *et al.* 1996, Sarell and McGuiness 1996, Walker 2004). Abundance of Brewer's sparrows may decline if sagebrush achieves greater than 50% cover (Dobler *et al.* 1996). In transition habitats between shrub-steppe and mixedgrass prairie, Brewer's sparrows prefer to nest in areas with lower sagebrush cover (5% to 10%), a denser understory of herbaceous species (graminoids and forbs) (64% to 73%), and shorter sagebrush plants (0.25 m to 1.0 m) than in shrub-steppe habitat (Feist 1968a, b; Best 1972, Walker 2004). In British Columbia, 20% of Brewer's sparrows were found to nest in dense sagebrush cover (*i.e.*, greater than 30%), 48% were found to nest in moderate sagebrush cover (*i.e.*,10% to 30%), and 32% were found to nest in sparse sagebrush cover (*i.e.*, less than 10%) (Sarell and McGuiness 1996).

In Wyoming, Ruehmann *et al.* (2011) compared Brewer's sparrow nest survival in native sagebrush prairie versus smooth brome-dominated plant communities (*i.e.*, modified grassland). The authors found that Brewer's sparrow nesting in smooth brome communities settled later, had smaller clutch sizes, higher nesting success, larger broods and higher nest survival rates than Brewer's sparrows nesting in native prairie. The authors suggest that the apparent benefit to Brewer's sparrows of smooth brome encroachment in this study may be related to increased nest concealment benefits and improved insect prey availability during the course of their study.

On a micro-habitat scale, Brewer's sparrows prefer to nest within sagebrush plants in shrub-steppe ecosystems in Idaho (Rich 1980, Reynolds 1981, Wiens and Rotenberry 1981). Shrub vigour appears to be an important component of shrub patches selected by Brewer's sparrows (Petersen and Best 1985, Knopf *et al.* 1990, Rotenberry and Wiens 1998). Big sagebrush shrubs that provide adequate concealment without overly dense foliage are preferred for nesting (Best 1972, Petersen and Best 1985). Sagebrush shrubs must be of sufficient height for Brewer's

sparrows to nest within them. Best (1972) found that shrubs with nests ranged from 28 cm to 63.5 cm tall in Montana. In Idaho, Petersen and Best (1985) found a mean nest shrub height of 69 ± 15 cm (range: 42 cm to 104 cm), whereas Rich (1980) observed a nest shrub height of 66.9 ± 11.3 cm. In terms of nest height, Best (1972) observed that nests were situated on average 16.5 cm above the ground surface (measured from the top of the nest); Petersen and Best (1985) observed an average height of 39 ± 10 cm (range: 20 cm to 50 cm); and Rich (1980) observed an average nest height of 28.2 ± 7.7 cm. In British Columbia, Sarell and McGuiness (1996) found that Brewer's sparrow nesting shrubs were, on average, 110 cm tall (range: 64 cm to 170 cm) and nests were placed, on average, at a height of 49 cm (range: 12 cm to 104 cm. Brewer's sparrows appear to nest higher in taller sagebrush shrubs (Rich 1980). Little information is known about Brewer's sparrow nesting requirements in Alberta. Semenchuk (1992) only notes that both subspecies of Brewer's sparrows in Alberta nest low to the ground in small shrubs.

1.3.3 Foraging Habitat

Wiens and Rotenberry (1998) studied foraging patch selection by Brewer's sparrows in the northern Great Basin in central Oregon. There, Brewer's sparrows selected shrub patches that were larger, more vigorous and dominated by big sagebrush. Brewer's sparrows also selected against foraging in shrub patches dominated by rubber rabbitbrush (*Ericameria nauseosa*) and green rabbitbrush (*Chrysothamnus viscidiflorus*). Larger, more vigorous shrub patches contain more arthropod prey items than smaller patches due simply to their larger size and more leafy material available for grazing insects (Wiens and Rotenberry 1998). The preference for big sagebrush over rabbitbrush likely represents a preference for the arthropod prey found on the former (Wiens and Rotenberry 1998). No information could be found on foraging requirements of Brewer's sparrows in Alberta.

1.3.4 Area Requirements

Reynolds (1981) observed an average territory size of 0.52 ± 0.15 ha for Brewer's sparrows in Idaho. Brewer's sparrow territory size in shrub-steppe habitat in Washington, Oregon and Nevada ranged from 0.55 ha to 2.36 ha (Wiens *et al.* 1985). Territory size was not related to any habitat features (*e.g.*, litter, grass, shrub or total vegetation cover) and was inversely related to breeding population density (Wiens *et al.* 1985). Territory sizes varied both within and between locations as well as between years for the same location, leading the authors to conclude that there may be no 'average' size of territory for the Brewer's sparrow (Wiens *et al.* 1985). Brewer's sparrow densities reported in the literature vary. Generally, densities are in the range of less than 1 pair/ha to 2 pairs/ha, although a maximum of 3 pairs/ha has been reported (Best 1972, Reynolds 1981, Schroeder and Sturges 1975, Rotenberry and Wiens 1980, Wiens and Rotenberry 1981, Wiens *et al.* 1985). In southeastern Alberta, densities of 1.1 pair/ha and 2.4 pair/ha were observed at the two sites monitored by Biermann *et al.* (1987).

1.4 Brewer's Sparrow Response to Grazing

Little research has been conducted on the effects of livestock grazing on Brewer's sparrows and/or their habitat. In general, grazing practices that promote shrub retention and growth in large, homogenous patches are likely to benefit Brewer's sparrows.

Data that does exist on the effects of livestock grazing on Brewer's sparrows is largely descriptive or correlative (Walker 2004). For example, in the Penticton-Osoyoos region of British Columbia, Sarell and McGuiness (1996) report that 0%, 31%, 55% and 14% of Brewer's sparrow breeding territories were located in Excellent, Good, Fair and Poor range condition classes, respectively. The authors suggest that the paucity of Brewer's sparrow nests in Excellent condition range may have been more a function of the lack of range in this condition class and not a presumed habitat preference. In Arizona, Bock *et al.* (1984) observed significantly higher numbers of Brewer's sparrows in winter in native prairie that had been continuously grazed compared to grassland that had been rested for more than 10 years. In Montana, Logan (2001) observed higher densities and higher nesting success by Brewer's sparrows in ungrazed plots than on adjacent grazed plots. In the shrub-steppe ecosystem of the northern Great Plains of the United States, Kantrud and Kologiski (1983) observed higher Brewer's sparrow abundance in lightly grazed fields compared to heavily grazed fields. In the transition zone between shrub-steppe and mixedgrass prairie, Brewer's sparrow abundance was higher in moderately grazed and lightly grazed fields compared to heavily grazed fields (Kantrud and Kologiski 1983).

Cattle will also directly impact Brewer's sparrow nests through trampling or dislodging (Walker 2004). However, the overall effect of livestock nest destruction and displacement on Brewer's sparrow survival is not known. Recent evidence from Grasslands National Park suggests that cattle have little impact on grassland bird nest survival (Lusk and Koper 2013).

In general terms, the effect of livestock grazing on grassland birds depends on the species (Fritcher *et al.* 2004). For example, certain species, such as grasshopper sparrows (*Ammodramus savannarum*) and bobolinks (*Dolichonyx oryzivorus*), prefer taller grass cover and decline in density as grazing intensity decreases. Other species, such as horned larks (*Eremophila alpestris*) and burrowing owls (*Athene cunicularia*), prefer shorter vegetation characteristic of early seral stages and heavier grazing intensity (Fritcher *et al.* 2004). Because of the different habitat preferences of grassland birds, habitat heterogeneity with a range of different grassland seral stages present is often advocated as a management strategy (Fuhlendorf *et al.* 2006). Continuous grazing with light to moderate grazing intensities has proven to be conducive to maintaining grassland bird populations in southern Saskatchewan (Lusk and Koper 2013). Applying a range of stocking rates over a given area will provide suitable habitat for multiple songbird species (Sliwinski and Koper 2015). Koper and Schmiegelow (2007) noted that grazing practices that promote taller vegetation but relatively short litter may benefit both songbirds and ducks.

Livestock grazing can also influence the composition of insect communities in grassland habitat, presumably thereby impacting the suitability of that habitat for Brewer's sparrows. However, the effect of livestock grazing on insects in unclear (Fleischner 1994). For example, Fielding and Brusven (1995) examined grasshopper densities in Idaho rangeland during drought conditions. They found that grasshopper densities were higher on rangeland that had been rested for 10 years or more compared to rangeland that was subject to rest-rotational grazing. In Arizona, Jepson-Innes and Bock (1989) found that in the summer, grasshopper density was 3.7 times greater on a site protected from livestock grazing compared to a site grazed by livestock, and in the fall, grasshopper density was 3.8 times greater on a site grazed by livestock compared to an ungrazed site. The reason for the apparent contradiction was that different subfamilies with different food preferences were dominant in different seasons. In Colorado, Welch *et al.* (1991) observed that

grasshoppers were significantly more abundant on a lightly grazed site than on a heavily grazed site. Because there was no difference between the same sites 19 years earlier, a long-term effect of livestock grazing was the likely cause. A number of European studies also suggest that high grazing intensity can lead to declines in arthropod richness and abundance (*e.g.*, Morris 1967).

1.5 Grazing Systems and Brewer's Sparrow Habitat Management

Little research has been conducted examining the effects of different grazing systems on Brewer's sparrow. Nevertheless, a few relevant studies have been completed, and with an understanding of the habitat needs of this species, general conclusions can be reached about the effects of different grazing systems. Various grazing systems and their potential effects on Brewer's sparrows are summarized in Table III-9. It should be stressed that there is no 'one-size-fits-all' approach or 'best' system that will work in all circumstances. Each ranch and lease has their own unique circumstances and will need to be approached differently. It is recommended to apply an adaptive approach that balances ranch/lease needs with the needs of Brewer's sparrows for certain habitat types (Adams *et al.* 2004). Grazing systems should be flexible and will ideally evolve over time to minimize their negative effects on Brewer's sparrows.

Table III-9 Grazing Systems and Brewer's Sparrow Habitat Management

Grazing System	Discussion
Continuous (Season-Long) Grazing	
Advantages:	Continuous grazing at light to moderate stocking rates has been shown to create a heterogeneous landscape with a mosaic of lightly to more heavily grazed plant communities. The effect of continuous grazing on sagebrush communities is generally unknown; some authors suggest that continuous grazing lead to an increase in sagebrush cover and others suggest sagebrush was always common on western rangelands (Miller *et al.* 1994, Peterson 1995). In Grasslands National Park, Ranellucci *et al*. (2011) observed higher grassland bird species richness and diversity in continuously grazed fields compared to twice-over rotationally grazed and ungrazed fields. A continuous grazing system in this mixedgrass prairie landscape led to the creation of a spatially heterogeneous landscape with a diversity of high and low use areas. A variety of vegetation types, with patches of short, grassy vegetation surrounded by sagebrush, will likely benefit Brewer's sparrows (Paige and Ritter 1999).
Disadvantages:	Continuous grazing can be detrimental to grassland bird habitat if stocking rates are high, resulting in uniform reduction in vegetation or litter cover. Continuous grazing could also lead to disturbance to Brewer's sparrow nests during the breeding season. Owens and Norton (1990, 1992) observed that livestock can have detrimental effects on sagebrush plants (particularly juvenile plants) through trampling under continuous grazing systems.

Rotational Grazing – Deferred Rotation	
Advantages:	Grazing that is deferred during the critical brood-rearing and nesting periods (*i.e.*, mid- to late summer) is thought to promote higher nest success among most ground-nesting birds (Holechek *et al.* 1982). Deferral minimizes loss of residual nesting cover early in the season and avoids trampling disturbance to nests. Koper and Schmiegelow (2007) noted that while deferred grazing may increase duck nesting success, the effect of this grazing system on most songbirds is less clearly beneficial, neither improving nesting success nor leading to negative effects. Deferred grazing will eliminate potential livestock disturbance of Brewer's sparrow nests if grazing commences after the breeding season. Deferred grazing also promotes improved range health by avoiding grazing during the vulnerable spring growth period.
Disadvantages:	The benefits of deferred grazing are reduced if heavy use occurs subsequent to deferral, thereby diminishing brood-rearing cover and reducing residual vegetation for the next nesting season (Holechek *et al.* 1982).
Complementary Grazing	
Advantages:	Complementary grazing allows for deferral of grazing in native prairie until after the critical spring growth period. Because native prairie is preferred over tame pasture for nesting, complementary grazing may keep livestock away from preferred, high quality Brewer's sparrow habitat during the nesting season. Although less preferred than native prairie, Brewer's sparrow will make use of tame pastures, provided there is sufficient shrub re-growth (McAdoo *et al.* 1989).
Disadvantages:	Early season use of tame pastures by livestock may lead to trampling of nests for birds that nest in tame pasture. Converting native prairie to tame pasture in order to develop a complementary grazing system is unlikely to benefit Brewer's sparrows due to the direct removal of the bird's preferred habitat type (McAdoo *et al.* 1989).
Rotational Grazing – Rest Rotation	
Advantages:	Rotational grazing systems allow fields to be rested in a sequential fashion for either an entire season or for part of a season (Holechek *et al.* 2003). This provides a mechanism to promote vegetation heterogeneity at a field level and allows areas to be free from livestock disturbance during the critical nesting and brood-rearing periods. Deferred and rest-rotation grazing may help to improve range health in Unhealthy fields and could also be used to minimize disturbance to Brewer's sparrow nests by livestock.
Disadvantages:	In Grasslands National Park, Ranellucci *et al*. (2011) observed higher grassland bird species richness and diversity in continuously grazed fields compared to twice-over rotational grazed as well as ungrazed fields.

	Continuous grazing in this study led to the creation spatially heterogeneous but temporally stable areas of high and low livestock use within fields, thus increasing diversity of microhabitats. Grazing intensities greater than 50% utilization may diminish the benefits of rotational grazing, leading to deteriorated range health, removal of residual nesting and brood-rearing cover and uniform vegetation structure (Kobriger 1980, Holechek *et al.* 1982). Fielding and Brusven (1995) found that grasshopper densities were higher on rangeland that had been rested for 10 years or more compared to rangeland subjected to rest-rotational grazing. Therefore, rest-rotational grazing may reduce the abundance of grasshoppers, a preferred food source of Brewer's sparrows (Best 1972, Peterson and Best 1986).

1.6 Beneficial Management Practice Recommendations

The following recommendations provide a variety of means by which to enhance Brewer's sparrow habitat in the Milk River and South Saskatchewan River watersheds and throughout the Grassland Natural Region as a whole. Provided below are both general recommendations as well as recommendations related to grazing management.

1.6.1 General Recommendations

Native Prairie Conservation

- Conserve remaining native prairie in southeastern Alberta, particularly those communities containing silver sagebrush. Brewer's sparrows are reliant on silver sagebrush for nesting (GoA 2016). Habitats with high biological diversity, a range of dense to sparse sagebrush patches, an intact understory of native graminoids and forbs and an intact cryptobiotic soil crust are preferred (Paige and Ritter 1999). Because of variability in Brewer's sparrow productivity between sites depending on the year, conservation of a range of sites spanning different environmental conditions is necessary (Mahony *et al.* 2006).
- Avoid converting native prairie to tame pasture or cropland due to the negative effects of such land conversion on Brewer's sparrows (McAdoo *et al.* 1989).
- Avoid habitat fragmentation in critical Brewer's sparrow habitat. Brewer's sparrows prefer large, contiguous tracts of spatially-similar native shrubland habitat (Knick and Rotenberry 1995).
- Discourage land management practices aimed at removing silver sagebrush from native plant communities. Killing or reducing cover of sagebrush plants with herbicide has been shown to have a negative effect on Brewer's sparrows in the United States (Schroeder and Sturges 1975, Kerley and Anderson 1995). Within Brewer's sparrow critical habitat, thinning of sagebrush stands may be necessary if cover reaches more than 50% (Paige and Ritter 1999). If thinning is necessary, care must be taken to avoid soil disturbance

and introduction of non-native plant species, and thinning work should be conducted when Brewer's sparrows are on their wintering grounds.

- Restore or reclaim cropland and tame pasture/hay fields to native grassland wherever possible.
- Ensure good riparian health along creeks in known Brewer's sparrow habitat. Active down-cutting of creek channels caused by poor riparian area grazing management will result in creeks being unable to access their floodplains and deposit sediment. Sedimentation appears to be important for maintaining silver sagebrush communities in southeastern Alberta (McNeil and Sawyer 2001).

Industrial Development Mitigation

- Reclaim or, preferably, restore abandoned wellsites in critical Brewer's sparrow habitat. Reclamation of native grassland should follow the provincial guidelines for native prairie reclamation (ESRD 2013). Natural recovery is preferable to seeding for silver sagebrush community reclamation, provided that erosion risks and invasion by non-native species are of low concern (Hickman 2010). Spatial planning tools such as the Grassland Vegetation Inventory (GVI) should be incorporated into all future energy developments in order to avoid new developments in silver sagebrush communities (Hickman 2010). Preferred native species and seeding rates should be specified in industrial development application documents if a company chooses against natural recovery (Hickman 2010). Certified weed-free native or wild harvested native seed should be required when seeding in intact silver sagebrush communities (Hickman 2010). Companies should be prohibited from seeding crested wheat grass and other invasive grass species during the reclamation/restoration activities (Hickman 2010).

Population Monitoring and Data Needs

- Develop and implement a standardized, long-term Brewer's sparrow population monitoring program. Ensure all observations are submitted for inclusion in the provincial Fish and Wildlife Management Information System (FWMIS). Consider reporting 'zero data' as well.
- Prepare and regularly update a provincial species at risk status report if the Brewer's sparrow becomes 'At Risk' or 'May Be At Risk'.

Pest Control

- Avoid the use of insecticides to control grasshoppers in known Brewer's sparrow habitat, if possible. Explore the use of non-toxic treatment options for killing unwanted pests.
- Manage livestock facilities near Brewer's sparrow habitat to reduce food resources for brown-headed cowbirds (Paige and Ritter 1999). For example, remove waste grain from corrals and feedlots and situate new livestock facilities in existing agricultural areas and not near sagebrush communities.

1.6.2 Grazing Recommendations

- Promote grazing practices that ensure the retention of silver sagebrush plant communities, including large, relatively homogenous, vigorous patches of sagebrush (Petersen and Best 1985, Knopf *et al.* 1990, Knick and Rotenberry 1995, Rotenberry and Wiens 1998). Continuous grazing using appropriate stocking rates should lead to the creation of habitat heterogeneity. Use of distribution tools (*e.g.*, salt, water development) will help to prevent over-use of sagebrush communities under continuous grazing.
- Use light to moderate grazing intensities (25% to 50% utilization) in known Brewer's sparrow habitat (Kantrud and Kologiski 1983, Adams *et al*. 2013a, b). Silver sagebrush appears to behave like a decreaser in southeastern Alberta, declining in cover in response to grazing, trampling and drought (Adams *et al.* 2004).
- Defer grazing on native grassland with breeding Brewer's sparrows until after the nesting season (end of May to mid-July) to avoid trampling or dislodging of nests by livestock. Deferred grazing will also benefit native grasses by avoiding defoliation during the critical spring growth period.

1.7 Research Recommendations

While much research has been completed on Brewer's sparrows to date, there are still a number of outstanding research questions. Most research has focused on aspects of the species biology, which are important first steps to managing the species. Additional research is needed on the effects of land management practices, such as livestock grazing, on the Brewer's sparrow. This research will ultimately benefit Brewer's sparrow conservation by increasing the knowledge of wildlife and land managers, who will then be able to adjust their management practices accordingly.

Specific research questions to be addressed with respect to the Brewer's sparrow include:

- Conduct a detailed study on Brewer's sparrow demographics in Alberta using scientifically rigorous methods.
- Identify and map preferred Brewer's sparrow habitat in southern Alberta. Relate preferred habitat characteristics to GVI ecological site types using resource selection functions, similar to the work completed by Williams *et al.* (2011). Much work has already been completed mapping silver sagebrush communities for greater sage-grouse (*Centrocercus urophasianus*) in southeastern Alberta.
- Determine Brewer's sparrow abundance in relation to range health of native prairie.
- Determine which grazing systems most benefit Brewer's sparrows and other grassland songbirds in the Milk River and South Saskatchewan River watersheds.

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II: MAMMALS

A. MOUSE GROUP (OLIVE-BACKED POCKET MOUSE AND WESTERN HARVEST MOUSE)

1 INTRODUCTION

The purpose of this report is to summarize the ecology and habitat requirements of the olive-backed pocket mouse (*Perognathus fasciatus*) and western harvest mouse (*Reithrodontomys megalotis*) in southern Alberta. Based on this information and supporting scientific studies, the potential effects of livestock grazing on these species and their habitats are discussed. Due to the limited information specific to grazing effects on these two mice species in Alberta, a comparative table of various grazing systems will not be given. The grazing response/ management discussion is followed by a summary of recommended beneficial management practices to enhance olive-backed pocket mouse habitat in the Milk River and South Saskatchewan River watersheds in Alberta, with broader application to the range of this species within the Grassland Natural Region of Alberta (Natural Regions Committee 2006). Lastly, a brief summary of research recommendations is provided.

2 OLIVE-BACKED POCKET MOUSE

2.1 Background

A member of the rodent family (Rodentia), the olive-backed pocket mouse is a small, solitary, nocturnal mouse that is predominantly found in mixedgrass and dry mixedgrass prairie in areas with loose, sandy soil (Gummer and Kissner 2004, Heisler *et al.* 2013). The species has short, dark, sandy-coloured dorsal pelage and white ventral pelage, with a thin, cream-coloured lateral line (Smith 1993). Although widely distributed in the arid grasslands of the Great Plains of the United States, the olive-backed pocket mouse is limited in its distribution to the southeastern corner of Alberta (Engley and Norton 2001). The species' range in Alberta extends north from the United States border to the confluence of the Bow and Oldman rivers and then northeast to the village of Empress along the Alberta-Saskatchewan border (Engley and Norton 2001. Olive-backed pocket mice also occur in south-central Saskatchewan and southwestern Manitoba (Manning and Jones 1988).

The olive-back pocket mouse is listed as 'Sensitive' under the Alberta General Status listing (Government of Alberta [GoA] 2016). The species is listed as a 'non-game animal' under the provincial *Wildlife Act*, making it illegal to kill, possess, buy or sell pocket mice. Nationally, the olive-backed pocket mouse has not been assessed by the Committee on the Status of Endangered Wildlife in Canada and is not listed under the federal *Species at Risk Act* (*SARA*) (COSEWIC 2016). Internationally, the olive-backed pocket mouse is listed as a species of 'Least Concern' by the International Union for Conservation of Nature and Natural Resources (IUCN) (Linzey and Hammerson 2008).

Currently, there are only a few local populations of olive-backed pocket mice in the province (GoA 2016). However, populations at these locations are considered to be dense (GoA 2016). There have only been four observations of this species in the vicinity of the Milk River (Gummer

and Kissner 2004). Due to their similar habitat requirements, the olive-backed pocket mouse often occurs in association with the Ord's kangaroo rat (*Dipodomys ordii*) in southeastern Alberta. However, there are no records of the Ord's kangaroo rat within the Milk River Watershed (Gummer 1997, Gummer and Robertson 2003). Suitable habitat for the Ord's kangaroo rat and the olive-backed pocket mouse may be declining in Alberta due to vegetation encroachment from altered grazing and fire regimes as well as other human land-use changes on the prairies, such as cultivation (Gummer 1997, Heisler *et al*. 2013). Further research is needed to determine the population status of the olive-backed pocket mouse in Alberta and to determine its response to land-use practices such as grazing.

2.2 Ecology

Pocket mice (*Perognathus* spp.), together with kangaroo rats (*Dipodomys* spp.) and kangaroo mice (*Microdipodops* spp.), belong to the heteromyid group of rodents. The kangaroo rat and the olive-backed pocket mouse are the only two heteromyid species found in Alberta (Gummer and Kissner 2004). Heteromyid rodents include some of the most specialized granivorous rodents in North American deserts (Fagerstone and Ramey 1996). These rodents feed predominately on seeds. They have distinctive external cheek pouches that are used to collect and transport large quantities of seeds (Manning and Jones 1988). Heteromyid rodents are thought to be important in seed dispersal and seedling establishment of certain native plants due to their seed caching behaviour (Fagerstone and Ramey 1996). Heteromyids harvest seeds well beyond their immediate food requirements and store the excess in caches located in burrows or in shallow pits dug into the soil surface (Price 1999). Stored seed caches provide nourishment for these rodents over the winter period, while excess seed in these caches serve as an important source of plant recruitment. Although seeds cached in burrows may be buried too deeply to germinate successfully, those buried near the surface are in a more favorable microsite for germination than seeds on top of the ground. Rodents also transport mycorrhizae associated with range plants, which may be important for establishing plant species on denuded range sites (Fagerstone and Ramey 1996). Pocket mice tend to influence the establishment of annual plants and small-seeded perennial grasses while kangaroo rats influence establishment of large-seeded plants (Fagerstone and Ramey 1996). Selective foraging by these rodents may also be important for stimulating plant species diversity due to feeding on competitively superior plants and their seeds.

All heteromyids are nocturnal and highly fossorial, spending much of their time in underground burrows and emerging to forage at night. Underground burrows are used for sleeping, shelter and birthing. Their nocturnal, fossorial lifestyle is well-suited to arid environments and predator avoidance. Olive-backed pocket mice, like Ord's kangaroo rats, tend to forage only during dark nights (with dim moonlight) to avoid being detected by predators (Gummer pers. comm.). Artificial lighting can therefore have negative implications for these species.

Olive-backed pocket mice are generally solitary animals (Smith 1993). These mice go into torpor in their burrows during the winter from mid-October to April, lowering their body temperature to conserve energy (Wrigley *et al.* 1991). Unlike true hibernators, they periodically arouse to feed on stored seeds and can be aroused in a few minutes when disturbed. Gestation periods of approximately one month have been reported for olive-backed pocket mice with breeding periods occurring from late April to August (Manning and Jones 1988, Wrigley *et al.* 1991). In

southern Manitoba, breeding begins in late April, with the first litter born in late May (Wrigley *et al.* 1991). A second breeding peak occurs from early July to mid-August (Wrigley *et al.* 1991). Litters may be produced once or twice a year with litter sizes ranging from two to 12 young (Manning and Jones 1988, Wrigley *et al.* 1991). Usually four to six young are born (Pattie and Fisher 1999). Young olive-backed pocket mice become sexually mature in the spring following their birth.

2.2.1 Diet

Olive-backed pocket mice are primarily granivorous and feed mainly on forb and grass seeds. Heteromyid rodents are known to harvest the seeds of a wide variety of plant species (Fagerstone and Ramey 1996). In southern Manitoba, olive-backed pocket mice feed on seeds from weedy forbs such as common pepper-grass (*Lepidium densiflorum*), wild buckwheat (*Fallopia convolvulus*) and Russian thistle (*Salsola kali*) (Wrigley *et al.* 1991). Olive-backed pocket mice in this case were found mainly at the edge of, or inside, crop or hay fields where weedy forbs were common (Wrigley *et al.* 1991). Other food items reported from cheek pouches of olive-backed pocket mice in Canada include: knotweed (*Polygonum* spp.), June grass (*Koeleria gracilis*), foxtail barley (*Hordeum jubatum*) and blue-eyed grass (*Sisyrinchium* spp.). These species are predominantly early successional species that colonize disturbed sites. Olive-backed pocket mice are also known to collect and eat small amounts of green vegetation and invertebrates (*e.g.*, grasshopper eggs) (Pattie and Fisher 1999). Like other heteromyids, pocket mice are adapted for water conservation and require little free water to drink, obtaining required water from their food (Fagerstone and Ramey 1996, Gummer 1997).

2.2.2 Predators

Common heteromyid rodent predators include owls, snakes, foxes, weasels (*Mustela* spp.) and American badgers (*Taxidea taxus*) (Gummer 1997).

2.3 Habitat Requirements

2.3.1 General

Olive-backed pocket mice are typically found in open, arid grasslands with loose, sandy soils, and sparse or very short vegetation (Manning and Jones 1988, Reynolds *et al.* 1999, Gummer and Robertson 2003, Heisler *et al.* 2013). Sites with low densities of shrubs may be selected to provide cover from raptors and other predators (Gummer and Kissner 2004). Wetlands and riparian habitats or other areas with taller cover or moderate to high shrub densities are usually avoided. Tall vegetation has been found to impede pocket mouse movement (Reichman and Price 1993). Riparian habitats also typically have less workable, fine textured soils. As olive-backed pocket mice use a hopping style of locomotion, they are able to move most easily in bare, open ground with sparse vegetation.

As part of the Multiple Species at Risk (MULTISAR) conservation program in southern Alberta, a Habitat Suitability Index (HSI) model was developed for olive-backed pocket mouse habitat use in Milk River Watershed (Gummer and Kissner 2004). The HSI model included five variables: soil texture; percentage of bare ground, graminoids and shrubs; and habitat type. As olive-backed pocket mice are restricted to grassland habitats, habitat type is the main

determining factor in this model. According to the model, grassland habitats best-suited to olive-backed pocket mice have moderate, moderately coarse or coarse textured soils, between 10% and 30% bare ground, between 60% and 80% graminoids and between 5% and 10% shrub cover.

2.3.2 Burrowing Habitat

Olive-backed pocket mice dwell in underground burrows that are 30 cm to 2 m below the soil surface and up to 6 m in diameter (Manning and Jones 1988). As these mice are not strong diggers they prefer loose, sandy soils that facilitate easy burrowing. Soils with moderate, moderately coarse or coarse textures are preferred (Gummer and Kissner 2004).

Salt (2000) reported an association between various mice and vole species, including olive-backed pocket mice, with northern pocket gopher (*Thomomys talpoides*) burrows in mixedgrass prairie in the Milk River Natural Area. The majority of mice or vole dens in this area were associated with old pocket gopher feeding tunnels. The larger mouse – gopher communities were inhabited by as many as four species, including deer mice (*Peromyscus maniculatus*), olive-backed pocket mice, sagebrush voles (*Lagurus curtatus*) and meadow voles (*Microtus pennsylvanicus*). Similar findings were reported by Wershler (2000) at the Antelope Creek Habitat Development Area, near Lake San Francisco, Alberta. In this case, however, voles and mice were reliant on old Richardson's ground squirrel (*Spermophilus richardsonii*) tunnels as pocket gophers were largely absent in the area.

2.3.3 Foraging Habitat

Pocket mice typically forage under perennial vegetation, which provides cover from predators and abundant seed resources (Price and Brown 1983). Olive-backed pocket mice are most abundant in areas with an intermediate amount of bare ground (approximately 20%) (Gummer and Kissner 2004).

2.3.4 Area Requirements

Olive-backed pocket mice are thought to have very small home ranges. Much of their time is spent in underground burrow complexes which are typically less than 10 m in diameter (Manning and Jones 1988). The largest documented movements of this species are less than 100 m (Prefaur and Hoffman 1975).

3 WESTERN HARVEST MOUSE

3.1 Background

Similar to the olive-back pocket mouse, the western harvest mouse is a member of the rodent family (Rodentia). It is nocturnal, very small, and has a long tail and naked ears. The western harvest mouse is one of five species of *Reithrodontomys* that occurs in the United States and Canada. Two subspecies of *Reithrodontomys megalotis* occur in Canada: *R. m. megalotis*, which occurs in British Columbia; and *R. m. dychei*, which occurs in Alberta (COSEWIC 2007). Although widely distributed in the central and western United States and Mexico, the western harvest mouse is limited in its distribution in Canada to southeastern Alberta and southern British

Columbia (COSEWIC 2007). This mouse species is predominantly found in dry mixedgrass prairie in dry shrub habitats with extensive vegetation cover such as tall grass and shrubs (Moulton *et al.* 1981, COSEWIC 2007).

The conservation status of the western harvest mouse is listed as 'Undetermined' under the Alberta General Status listing (GoA 2016). Similar to the olive-backed pocket mouse, the species is listed as a 'non-game animal' under the Alberta *Wildlife Act.* The western harvest mouse was assessed by COSEWIC in 1994 and given a status of 'Data Deficient' (Nagorsen 1994). The species was re-assessed by COSEWIC in 2007 and given an updated status of 'Endangered', a status it retains to this date (COSEWIC 2007, 2016). The western harvest mouse was added to Schedule 1 of *SARA* in 2009, thereby giving the species federal protection. Internationally, the western harvest mouse is considered a species of 'Least Concern' by the IUCN (Linzey and Matson 2008).

The western harvest mouse has only been observed in four locations in Alberta, including near the Milk River, Medicine Hat, Manyberries, and along the North Saskatchewan River in the Suffield National Wildlife Area (SNWA). The observations in SNWA were the only records in Alberta after the 1960s, and occurred in the mid-1990s (Reynolds *et al.* 1999, COSEWIC 2007). This population may be disjunct from southern populations in Montana. Further research is needed to determine the population status of the western harvest mouse in Alberta and to determine its response to land-use practices such as grazing.

3.2 Ecology

The western harvest mouse is in the Neotominae subfamily, which includes 16 genera and is comprised of mice, packrats and woodrats. Western harvest mice are one of five members of the *Reithrodontomys* genus which are endemic to North America, but it is the only species in Canada. Neotomine rodents occur throughout North America in a variety of habitats. Neotomine rodents vary from herbivores to carnivores (Nowak 1999). Western harvest mice are omnivorous and feed on seeds, new vegetation growth and invertebrates (COSEWIC 2007). Neotomines can be nocturnal, diurnal or crepuscular (Nowak 1999). Western harvest mice are nocturnal and live above-ground (COSEWIC 2007).

Although observed year-round, it is likely these mice go into torpor during the winter months by lowering their body temperature to conserve energy. Unlike true hibernators, they would periodically arouse to feed on stored seeds. Gestation periods of approximately 21 to 24 days have been reported for western harvest mice, with breeding occurring from March to November in the United States (Whitaker and Mumford 1972). Breeding in Alberta is likely a shortened period based on the shorter growing season and colder climate. A maximum of four or five litters may be produced each year, with litter sizes ranging from one to seven young (Hayssen *et al.* 1993, Nowak 1999). Young western harvest mice become sexually mature four to 12 months following their birth.

3.2.1 Diet

The western harvest mouse is omnivorous and feeds on seeds from grasses, legumes and other forbs; new vegetation growth; and invertebrates such as arthropods and lepidopteran larvae (COSEWIC 2007, Whitaker and Mumford 1972). It appears that the western harvest mouse does not forage on woody plants but will forage on cereal crops, such as rye (*Secale cereale*), when homogenous crops are dominant (Whitaker and Mumford 1972).

3.2.2 Predators

Potential predators of western harvest mice include owls, hawks, jays, loggerhead shrike (*Lanius ludovicianus*), prairie rattlesnake (*Crotalus viridis*), foxes, weasels (*Mustel*a spp.), American badgers (*Taxidea taxus*) and skunks (summarized by COSEWIC 2007).

3.3 Habitat Requirements

3.3.1 General

The western harvest mouse is found primarily in dry mixed grass prairie in dry shrub habitats (COSEWIC 2007). Western harvest mice are associated with extensive vegetation cover, such as tall grass and shrubs, and appear to avoid areas with low vegetation and low litter cover (Agnew *et al.* 1986). Western harvest mice exhibit a strong affinity for grass litter (Clark and Kaufman 1991).

3.3.2 Nesting Habitat

Nests are spherical or cup-shaped and are typically located in shrubs located up to 1 m off the ground (Webster and Jones 1982). Burrows or ground nests are also occasionally used (Birkenholtz 1967). Nests are composed of grasses and woody vegetation on the exterior and soft vegetation or fluff on the interior (Wilson and Ruff 1999). Nests have been located within patches with moderate litter (Clark and Kaufman 1991). Therefore, western harvest mice not only forage but also nest in areas with well-developed litter. However, as Clark and Kaufman (1991) showed, although there is an affinity for litter such that foraging and nesting in areas with moderate litter is preferred over areas with low litter, there is no indication that they do not select for areas with high litter.

3.3.3 Foraging Habitat

Western harvest mice have foraged selectively in patches with moderate litter (Clark and Kaufman 1991). They avoid areas that have been hayed or grazed, which suggests an association with mixedgrass prairie that is not heavily disturbed (Kaufman and Kaufman 2008).

3.3.4 Area Requirements

Home ranges appear to be between 0.44 ha to 1.12 ha (Meserve 1977, O'Farrell 1978). Dispersal by western harvest mice has been documented in Indiana where they have expanded their range over 25 years (Whitaker and Mumford 1972, Leibacher and Whitaker 1999). In Indiana, it appears the species invades early seral habitats and establishes a large population and over the 25 years, the species has dispersed up to 160 km.

4 GRAZING AND MICE

Vegetation structure is thought to play a major role in determining the general composition of grassland small mammal communities (Grant *et al.* 1982). Studies have found discernible differences between species diversity, total small mammal biomass and community composition along a reduced vegetation cover gradient from tallgrass to mid to shortgrass prairie (Grant *et al.* 1982). Precipitation, grazing and fire are the predominant factors that determine the structure and function of grasslands (Jones *et al.* 2003). In general, small mammals found in taller grass communities, such as the western harvest mouse, are more markedly influenced by reduced vegetation cover than small mammals, such as the olive-backed pocket mice or the Ord's kangaroo rat, that are adapted to short or sparse vegetation (Grant *et al.* 1982).

In addition to influencing vegetation cover, domestic or wild ungulate grazing affects plant species composition and diversity, primary productivity, standing crop biomass, soil compaction and soil moisture attributes (Grant *et al.* 1982). These attributes can impact the availability of food and the stability of small mammal burrows. Grazing has the potential to negatively impact heteromyid rodents due to direct trampling of burrows and soil compaction or by removing seed heads (Heske and Campbell 1991, Gummer pers. comm.). Grazing may benefit olive-backed pocket mice and other seed-eaters if it results in an increase in the abundance of annual grasses, forbs or shrubs that produce more seeds than perennial grasses (Grant *et al.* 1982, Jones and Longland 1999). Studies have found that livestock grazing was important for increasing forb species richness in mixedgrass prairie in southwestern Saskatchewan (Thorpe and Godwin 2003); however, grazing has the potential to negatively impact rodents such as Neotomines by reducing food availability through competition with grazing livestock or reduced seed production, as well as reduced cover for predator avoidance and thermoregulation (Johnson and Horn 2008).

Jones *et al.* (2003) compared small mammal communities between grazed and ungrazed grassland in southeastern Arizona. Ungrazed sites had been excluded from cattle grazing for 33 years. According to the results of this study, heteromyid rodents as a group were more common on grazed plots and were positively correlated with bare ground cover. In contrast, *Reithrodontomys* spp. were significantly more common on ungrazed plots with taller vegetation cover.

4.1 Olive-Backed Pocket Mouse Response to Grazing

Few studies have specifically examined the response of olive-backed pocket mice to livestock grazing in Alberta or elsewhere in its range. The pocket mice of the Great Plains, as a group, show varying responses to grazing (Fagerstone and Ramey 1996). Several species, such as the Great Basin (*Perognathus parvus*), Price (*P. penicillatus*) and Bailey (*P. baileyi*) pocket mice, prefer heavier protective cover of either perennial grasses or shrubs and are negatively impacted by heavy grazing. In contrast, the Arizona pocket mouse (*P. amplus*), silky pocket mouse (*P. flavus*) and hispid pocket mouse (*P. hispidus)* are found in more open areas with sparse grass cover, and respond favourably to grazing (Fagerstone and Ramey 1996). As olive-backed pocket mice prefer open habitats with sparse and short vegetation and moderate amounts of bare ground, it is likely that grazing will similarly benefit this species (Gummer pers. comm.). Grazing may be

particularly beneficial if it contributes to maintaining active sand dune complexes that are important to olive-backed pocket mice and Ord's kangaroo rats in Alberta. Trampling and grazing along sandy formations can lead to localized blowouts and reactivation of sand dune complexes (Houston 2000). Blowouts and erosion form along well-used paths and areas with heavily reduced vegetation cover. Localized erosion also results from heavy livestock use points such as at water sources or salting stations. In recent years, Ord's kangaroo rats have been found to be increasingly abundant on roads, fireguards and other human-disturbed areas in the sandhills in the northern portion of the SNWA (Gummer 1997, Gummer pers. comm.). This area has been excluded from grazing since the 1970's, a factor which is thought to have contributed to the apparent stabilization of sand dunes in the area and which may explain the shift in habitat use by kangaroo rats from natural habitats to human-modified areas (Gummer pers. comm.). A similar habitat shift may be occurring with olive-backed pocket mice, although further research is required. Due to their smaller home ranges, olive-backed pocket mice require smaller patch sizes of suitable habitat than Ord's kangaroo rats.

4.2 Western Harvest Mouse Response to Grazing

Studies assessing the response of western harvest mice to grazing have been conducted in the United States, often in conjunction with the study of the response of other rodents. Western harvest mice have a preference for habitats with high vegetation cover. Livestock grazing decreases the amount of understory vegetation, cover and food for small mammals. Therefore, grazing impacts habitats and affects small mammal associations (Moulton *et al.* 1981, COSEWIC 2007). The reduction in organic litter due to grazing may also impact western harvest mice, since this species has a preference for specific amounts of litter (Clark and Kaufman 1991). Western harvest mice selected grasslands where feral horse grazing had been absent for 10 to 14 years over actively grazed grasslands (Beever and Brussard 2004). In addition, Kaufman and Kaufman (2008) showed avoidance, or near avoidance, of hayed rangelands (native and tame) by western harvest mice due to a reduction in standing vegetation and litter cover. It has been suggested that the presence of western harvest mice in SNWA is likely because there has been little cattle grazing in the area compared to surrounding areas (COSEWIC 2007).

5 GRAZING SYSTEMS AND MOUSE HABITAT MANAGEMENT

5.1 Olive-Backed Pocket Mouse

In general, conservative stocking rates are suggested for sandy ecological sites due to the greater potential for erosion (Houston pers. comm.). Once-over grazing, whereby pastures are grazed only once during a single season, is also recommended (Houston pers. comm.). These recommendations are geared toward maintaining sustainable forage productivity and adequate carryover of organic residue (litter) but may not allow for adequate disturbance to maintain active dunes and areas of short and sparse vegetation necessary to sustain high densities of olive-backed pocket mice and other rare species. The decision to locally increase stocking rates or apply twice-over grazing should first consider the needs of other priority management species and the potential economic impacts on the ranching operation. Because olive-backed pocket mice have small home ranges and occupy burrow complexes less than 10 m in diameter, only small patches of suitable habitat are required. Thus, conservative stocking rates may be

appropriate for maintaining habitat for this species where livestock distribution tactics are used effectively to shift patches of heavy use over the landscape. Shifting areas of high livestock use followed by periodic rest can create localized areas suitable for colonization by olive-backed pocket mice.

Deferred spring grazing may be beneficial to olive-backed pocket mice as it allows for improved seed production, seedling establishment and restored plant vigour (Adams *et al.* 1991). This has benefits in terms of improving food supplies available to pocket mice. Deferral requires ranchers to have alternate forage sources in the spring, such as the use of tame pastures through complementary grazing. Deferral can also be achieved using deferred rotation grazing whereby spring use is alternated between multiple native fields from year to year (Adams *et al.* 2004). Season-long (or continuous) grazing may also be favourable for pocket mice if it creates a diversity of patch types with variable vegetation canopy, litter cover and exposed soil due to variable grazing pressure. Light to moderate stocking rates as well as appropriate use of livestock distribution tools help to promote patch diversity under a season-long grazing system (Adams *et al.* 2004). Season-long grazing under heavy stocking rates tends to promote large areas of heavy use and may be detrimental to promoting seed production.

5.2 Western Harvest Mouse

The maintenance of good vegetation cover and moderate litter appears to be important criteria for optimal western harvest mouse habitat. No to light grazing is recommended for areas where western harvest mice are present. Light grazing under conservative stocking rates will maintain vegetation cover and structure and have less impact on litter abundance than higher stocking rates. Until the definite effects of livestock grazing on western harvest mice are known, it is recommended that, where possible, no grazing occur, such as in the SNWA.

6 BENEFICIAL MANAGEMENT PRACTICE RECOMMENDATIONS

The maintenance and appropriate management of sandhill, sand plain and sand dune habitats is important for olive-backed pocket mice, as is the amount of vegetation and litter cover in native prairie for western harvest mice within the in the Milk River and South Saskatchewan River Watersheds. Sandhill habitats are fragile ecosystems that are sensitive to disturbance and not suited to cultivation.

The following general land-use and grazing recommendations provide a variety of means by which to protect or maintain suitable habitat for the olive-backed pocket mouse and western harvest mouse within the Milk River and South Saskatchewan River Watersheds and throughout the Grassland Natural Region of Alberta. Further research is required to improve our understanding of these species in order to evaluate or refine appropriate management recommendations (see Section 7).

6.1 General Recommendations

6.1.1 Olive-backed Pocket Mouse

- Conserve existing native prairie sandhill or sandy ecological site habitats from cultivation or extensive development. Although cultivated areas may be used by pocket mice, these areas do not provide secure habitats due to the potential for burrows to be destroyed by farm machinery. Cultivated areas also pose risks to mice associated with herbicide or pesticide use.
- Avoid the use of herbicides or pesticides near known olive-backed pocket mouse habitats due to potential for direct poisoning or consumption of contaminated grains by pocket mice.
- Avoid widespread poisoning programs involving distribution of poisoned grain such as for Richardson's ground squirrel control. Localized areas of disturbance created by Richardson's ground squirrels provide important habitat for olive-backed pocket mice (Salt 2000, Wershler 2000, Gummer pers. comm.). Pocket mice and other granivorous rodents are also vulnerable to consumption of poisoned grain.
- Maintain low to moderate (5% to 10%) shrub cover in pocket mice habitats (Gummer and Kissner 2004). Shrubs are important for providing cover from predators and also serve as important seed sources.
- Where possible, conduct pre-development surveys to locate potential olive-backed pocket mouse burrows. Use the HSI map to map out high priority areas to survey or to avoid during pre-development planning and pipeline routing. Spot-lighting or live-trapping can be used to survey for pocket mice (Gummer pers. comm.). Olive-backed pocket mice surveys may help to locate otherwise undetected populations of Ord's kangaroo rats within the Milk River and South Saskatchewan River watersheds due to their similar habitat preferences (Gummer pers. comm.).
- Minimize artificial lighting in olive-backed pocket mice habitat from facilities or due to night-time drilling or pipeline construction where bright lighting is required (Gummer pers. comm.). Olive-backed pocket mice, like Ord's kangaroo rats, tend to forage only during dark nights and minimize their above-ground activity during relatively bright night-time conditions (*e.g.*, moonlight) (Gummer 1997, Gummer pers. comm.). This behaviour is thought to be a predator avoidance strategy. Therefore, these rodents have a limited number of suitable foraging nights available during periods of seed availability in which to gather the resources required to survive the six to eight month hibernation period.
- Avoid night-time traffic or night-time construction activities in sandhill type areas where olive-backed pocket mice are likely to occur (Gummer pers. comm.). Olive-backed pocket mice are nocturnal and may be disturbed by night-time activities.
- Pipeline construction during hibernation (*i.e.*, from October to April) is preferred, when burrows are not directly impacted (Gummer pers. comm.).
- Avoid leaving open trenches (*e.g.*, from pipelines) in pocket mouse habitats during the spring and summer months (Gummer pers. comm.). Trenches create a trap for both mice and snakes, preventing escape and leaving mice vulnerable to predation.

- Avoid heavy, uncontrolled off-road vehicle traffic through areas with potential olive-backed pocket mice burrows to avoid burrow destruction.

6.1.2 Western Harvest Mouse

- Protect existing native prairie habitats from cultivation or extensive development. Although cultivated areas have been used by western harvest mice, these areas do not provide secure habitats for foraging or nesting. Cultivated areas also pose risks to western harvest mice due to adverse effects associated with herbicide or pesticide use.
- Avoid use of herbicides or pesticides near known western harvest mouse habitats due to potential for direct poisoning or consumption of contaminated grains by mice.
- Avoid widespread poisoning programs involving distribution of poisoned grain such as for Richardson's ground squirrel control. Western harvest mice and other rodents are also vulnerable to consumption of poisoned grain.
- Maintain good shrub cover in western harvest mice habitats. Shrub cover is important for providing cover from predators, nesting locations and seed sources (COSEWIC 2007).
- Avoid night-time traffic or night-time construction activity in areas where western harvest mice are likely to occur.
- Minimize artificial lighting in western harvest mice habitat from oil and gas facilities or night-time drilling or pipeline construction where bright lighting is required. Similar to olive-backed pocket mice, western harvest mice are nocturnal and may be adversely affected by bright lights.
- Avoid leaving open trenches (*e.g.*, from pipelines) in western harvest mouse habitats during the spring and summer months.
- Avoid haying roadsides near critical western harvest mouse habitat (Kaufman and Kaufman 2008).

6.2 Grazing Recommendations:

6.2.1 Olive-backed Pocket Mouse

Properly managed livestock grazing is often considered the most sustainable use of sandhill or sand plain habitats (Houston 2000). Some degree of grazing may indeed be required to maintain active sand dune complexes (Houston 2000). Grazing is also important for stimulating plant species diversity and creating patches of short vegetation and reduced litter that are suitable for olive-backed pocket mice. A range of grazing intensities is likely preferable to ensure a stable seed source, while creating locally small areas with sparse vegetation and patches of bare soil. Appropriate grazing systems for olive-backed pocket mice should be developed site-specifically for ranches participating in southern Alberta conservation programs.

To promote habitat for olive-backed pocket mice, the following general grazing practices are recommended:

- Promote deferred spring grazing through the use of either complementary or deferred rotation grazing to allow seed set and improved plant vigour.

- Promote patchy grazing with a gradation of light to heavily grazed patches across the landscape. This may be achieved through an extensive grazing system utilizing light to moderate stocking rates and appropriate livestock distribution practices.
- Apply distribution practices such as herding and salt or water placement to create habitat patchiness across the landscape. Allow for periodic rest of highly used areas. Small patches with 10% to 30% bare ground, 60% to 80% graminoids and 5% to 10% shrub cover will provide suitable habitat for olive-backed pocket mice (Gummer and Kissner 2004).

6.2.2 Western Harvest Mouse

Although the level of grazing intensity that is tolerated by the western harvest mouse is unknown, the species appears to respond negatively to grazing and plant defoliation. Therefore, grazing in areas with known or potential western harvest mouse populations should be cautiously implemented. If grazing does occur in western harvest mouse habitat, it should be light in intensity. Appropriate grazing systems for western harvest mice should be developed site-specifically for ranches participating in the MULTISAR conservation program.

To promote habitat for western harvest mice, the following general grazing principles apply:

- Use conservative stocking rates. Conservative stocking rates, such as those recommended in the Dry Mixedgrass and Mixedgrass subregion range plant community guides (Adams *et al.* 2013a, b), will promote high vegetation cover and good litter cover across the landscape.
- Livestock distribution tools, such as strategic water developments, will also help to prevent over-use of preferred areas and ensure key western harvest mouse habitat is not adversely affected by livestock.
- If possible, fence off and avoid grazing critical western harvest mouse habitat, at least until the exact effects of livestock grazing on the species are known.

7 RESEARCH RECOMMENDATIONS

The scarcity of olive-backed pocket mouse and western harvest mouse records within the Milk River and South Saskatchewan River watersheds underscores the need for further research and population surveys to be conducted in this region. Population surveys could be designed to provide more information about plant community types that olive-backed pocket mice and western harvest mice are associated with, their relationship with range health and their association with either natural or human-modified habitats.

Additional research needed for the olive-backed pocket mouse and western harvest mouse include:

- Information on the level of grazing intensity and type of grazing system(s) best suited for creating and maintaining habitat for these mice species while simultaneously maintaining rangeland productivity.

- Research the use of controlled grazing or prescribed burning as mechanisms to maintain the structure and function of active sand dune complexes for rare flora and fauna (Houston 2000).
- Conduct more in-depth studies of olive-backed pocket mice and western harvest mice survival, reproduction, hibernation patterns, home range sizes and seasonal movements, especially in Canada (Gummer pers. comm.).
- Further examine olive-backed pocket mouse associations with larger burrowing mammals, such as northern pocket gophers or Richardson's ground squirrels (Salt 2000, Wershler 2000). Ongoing studies of Ord's kangaroo rats in Canadian Forces Base (C.F.B.) Suffield continue to provide the opportunity to collect incidental information about olive-backed pocket mice. An assessment of the potential effects of kangaroo rat occupation of human-modified areas at C.F.B. Suffield may have particular relevance to pocket mouse habitat management in the Milk River and South Saskatchewan River watersheds. Further study of the western harvest mouse population in SNWA may provide additional information on this mouse species in Canada and help identify what factors are impacting its habitat and population.
- Revise the HSI map for the olive-backed pocket mouse as new data and Geographic Information System (GIS) layers become available.

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B. MAMMAL BURROW GROUP (SWIFT FOX, AMERICAN BADGER AND RICHARDSON'S GROUND SQUIRREL)

1 INTRODUCTION

The purpose of this report is to summarize and compare the ecology and habitat requirements of the swift fox (*Vulpes velox*), American badger (*Taxidea taxus taxus*) and Richardson's ground squirrel (*Spermophilus richardsonii*) in southern Alberta. Based on this information, livestock grazing interactions and the potential effects of various grazing systems on these species and their habitat are discussed. This discussion is followed by a summary of recommended beneficial management practices (BMPs) to enhance habitat for these species in the Milk River and South Saskatchewan River watersheds in Alberta and throughout their range in the Grassland Natural Region of Alberta (Natural Regions Committee [NRC] 2006). Lastly, a brief summary of additional research needs is presented.

The swift fox, American badger and Richardson's ground squirrel have similar habitat preferences and are ecologically linked due to their mutual reliance on underground burrows for breeding, shelter and predator avoidance. Richardson's ground squirrels are also an important prey item for badgers and, to a lesser extent, for swift fox. Historically, these three mammals coexisted with bison (*Bison bison*) throughout their range across the plains of North America. Presently, livestock grazing is the dominant land-use in the native prairie habitats occupied by these species in southern Alberta. In addition to grazing management, it is also important to consider strategies to minimize potential impacts due to other human activities and land-use practices which can pose a risk to these species or their habitats.

2 SWIFT FOX

2.1 Background

The swift fox, North America's smallest canid, is associated with the Great Plains and was once common throughout the short and mixedgrass prairie regions of the continent (Cotterill 1997, Moehrenschlager and Moehrenschlager 2001, Schauster *et al.* 2002). The species' historical range coincided with that of the bison, encompassing an estimated 1.5 million km^2 of land in Canada and the United States (Smeeton and Weagle 2000, Sovada *et al.* 2009). The swift fox is presently found in approximately 44% of its historical range in the United States and 3% of its historical range in Canada (Sovada *et al.* 2009). In Canada, the species' current range is limited to two populations, one along the Alberta/Saskatchewan/Montana border and a second, smaller pocket located in Grasslands National Park in Saskatchewan (Cotterill 1997). The species current extent of occurrence in Canada is 21,544 km^2 (Alberta Swift Fox Recovery Team [ASFRT] 2007). In the United States, there are currently low abundances of swift fox in Montana, Nebraska and South Dakota; higher abundances occur in Colorado, Kansas, and Wyoming, and in parts of Texas, New Mexico and Oklahoma (Cotterill 1997, Schauster *et al.* 2002).

The swift fox is currently listed as 'At Risk' under the Alberta General Status listing (Government of Alberta [GoA] 2016). It has been designated as an 'Endangered' species under the provincial *Wildlife Act*. The *Act* contains provisions for the protection of swift fox dens. Nationally, the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) officially designated the swift fox as 'Extirpated' in Canada in 1978 (Saskatchewan Department of Tourism and Renewable Resources 1978). A status re-examination by COSEWIC in 1998 led to the downlisting of the species to 'Endangered' status, which was re-confirmed in 2000 (Carbyn 1998, COSEWIC 2000a). The most recent status re-examination in 2009 led to a further downlisting to 'Threatened' status (COSEWIC 2009). The swift fox is currently listed as a 'Threatened' species under Schedule 1 of the *Species at Risk Act* (*SARA*). Internationally, the International Union for Conservation of Nature and Natural Resources (IUCN) has designated the swift fox as a species of 'Least Concern' (Moehrenschlager *et al.* 2008).

The swift fox experienced dramatic range-wide declines in the late 1800s and early 1900s, leading to their extirpation from Canada and northern Montana by the late 1930s (Cotterill 1997, Moehrenschlager and Moehrenschlager 2001). The arrival of European settlers and the concurrent extensive loss of native prairie habitat and alteration of prairie ecosystems as well as unregulated trapping, shooting and incidental poisoning are thought to be the primary factors responsible for the swift fox population crash (Carbyn 1995, Brechtel *et al.* 1996, Cotterill 1997, Moehrenschlager and Moehrenschlager 2001).

Concerns over the apparent extirpation of the swift fox in Canada led to efforts to reintroduce the species back into southern Canada. The Swift Fox Reintroduction Program began in Canada in 1983 and continued until 1997 (Moehrenschlager and Moehrenschlager 2001). Swift fox recovery efforts in Canada were spearheaded by Miles and Beryl Smeeton, owners of the Cochrane Ecological Institute, located near Cochrane, Alberta. A total of 841 captive-raised swift foxes and 91 translocated swift foxes were released into southeastern Alberta and southwestern Saskatchewan during the course of the program (Smeeton and Weagle 2000). Areas chosen for swift fox reintroduction were mainly composed of grazing lands and mixedgrass prairie (Herrero *et al.* 1986). The Swift Fox Recovery Team was established in Canada in 1989 to guide conservation and recovery efforts for the species.

In 1989, a reintroduction was also attempted in the Milk River Ridge area of south-central Alberta with the release of 61 swift fox individuals (Moehrenschlager and Moehrenschlager 2001). Reintroduction into the Milk River Ridge region was discontinued due to a rabies outbreak in the striped skunk (*Mephitis mephitis*) population as well as due to a high abundance of predators. The area is not thought to contain a successful breeding population of swift foxes at this time (COSEWIC 2009).

In 2004, 15 swift fox were released on to the Blood Indian Reserve in southwestern Alberta (Smeeton 2006). The success of this program is not known, although of the five radio-collared individuals tracked, three went missing, one shed its collar and one was found dead (COSEWIC 2009). Future reintroductions in this area are being considered (ASFRT 2007).

Three censuses have been conducted for swift fox in Canada since 1996. Following reintroduction efforts, a winter census conducted from 1996 to 1997 estimated the Canadian

population of swift foxes to be approximately 289 foxes (Cotterill 1997). A census of populations in Alberta, Saskatchewan and Montana conducted from 2000 to 2001 revealed that the swift fox population had increased over a four-year span without the supplementation of additional swift fox releases (Moehrenschlager and Moehrenschlager 2001). The Canadian population of foxes was estimated to have increased from 289 foxes to 656 foxes as a result of the reintroduction program (Moehrenschlager and Moehrenschlager 2001). A third census of swift fox conducted in Canada and Montana in 2005 and 2006 found an estimated 647 foxes in Canada (Moehrenschlager and Moehrenschlager 2006). Other findings from the 2005/2006 survey included: population connectivity had improved, with no more than one township between swift fox sightings; and the majority of the range expansion of the species occurred in northern Montana. The results from the Canadian surveys are supported by surveys elsewhere, which suggest the swift fox is expanding its range and populations size in North America (Sovada *et al.* 2009). Although the results of the Alberta surveys are encouraging, it has been debated as to how many individuals are needed to maintain a self-sustaining, viable population with sufficient genetic variability (Moehrenschlager and Moehrenschlager 2001). Recent evidence of long-distance dispersal between populations once thought to be isolated from each other suggests that swift fox in Canada may now be part of one large metapopulation with swift fox in Montana (Ausband and Moehrenschlager 2009). Thus, fears of genetic isolation may be unwarranted.

Several factors limit the long-term re-establishment of the swift fox in Alberta and Saskatchewan, including: vulnerability to poisoning (rodenticides and pesticides); predator control programs; predation; interspecific competition with coyotes (*Canis latrans*) and red foxes (*Vulpes vulpes*); a changing prey base; canid disease; continuing habitat fragmentation and habitat loss; increasing agricultural, urban, and industrial development; and road mortalities (Carbyn 1995, Cotterill 1997, Sovada *et al.* 1998). Due to its small size, the Canadian swift fox population is also vulnerable to climatic conditions such as drought and severe winters (Cotterill 1997).

2.2 Ecology

The breeding season for swift fox usually begins in late January, with pups produced from late April to early May (Cotterill 1997, Pattie and Fisher 1999). Moehrenschlager and Macdonald (2003) suggested that breeding does not begin until mid-February. The preliminaries of courting can take a long time for newly paired swift foxes as they are ‘fastidious’ in mate selection (Weagle and Smeeton 1995). Swift fox appear to be monogamous and will establish a pair bond if they are successful breeders (Scott-Brown *et al.* 1986). Litter sizes range from three to eight young, with four being the most common number (Moehrenschlager 2000, Olson and Lindzey 2002a). Prey abundance is thought to influence swift fox productivity (Olson and Lindzey 2002a). Breeding may begin within the first year of age, with the first breeding season occurring later than usual (Cotterill 1997). During breeding, activity centers on the natal den (Alberta Fish and Wildlife Division (FWD) 1990). Newborn pups depend on their parents for food and protection (FWD 1990). Initially, females remain underground with the pups while male swift foxes hunt and bring food to the den. Pups are weaned at six to seven weeks but will stay near the den and with the adults for several months (FWD 1990). Juvenile swift foxes will typically disperse in the late summer and autumn when they reach four to five months of age (Cotterill

1997). It is at this time of year that swift fox relocation efforts have proven most successful (Smeeton and Weagle 2000).

Swift fox are generally more active at night; however, their activity patterns can vary seasonally (Smeeton and Weagle 2000). Moehrenschlager *et al.* (2004) suggested that swift foxes are largely nocturnal in winter, but become more crepuscular in summer. Thompson and Gese (2012) suggested that swift fox behaviour in winter is governed by prey acquisition, whereas their behaviour the rest of the year is governed by security from intraguild predation. Swift fox in captivity can survive up to 14 years; however, the lifespan of swift foxes in the wild is typically between three to seven years (FWD 1990, Reid 2006). Moehrenschlager *et al.* (2007a) reported annual adult survival rates of 0.38 to 0.63 and annual juvenile survival rates of 0 to 0.50. Annual mortality rates of swift fox reported in the literature for other parts of the species' range include 0.40 to 0.69 in Wyoming (Olsen and Lindzey (2002a) and 0.60 to 0.88 (Thompson *et al.* 2008) and 0.50 to 0.92 (Thomson and Gese 2012) in Colorado. Predation is primary cause of swift fox mortality (Moehrenschlager *et al.* 2007a). Swift foxes are very territorial, with substantial overlap of home ranges and core areas between members of breeding pairs, but little to no overlap between non-breeding neighbours (Lebsock *et al.* 2012).

2.2.1 Diet

Swift foxes are nocturnal, opportunistic feeders whose food habits vary seasonally and consist of a variety of prey, including mammals, carrion, invertebrates, vegetation (including seeds, berries and grass), amphibians, reptiles and small birds (Cutter 1958a, Carbyn 1995, Brechtel *et al.* 1996, Cotterill 1997, Sovada *et al.* 2001). In northern ranges, the most important source of food, particularly during the winter, is thought to be small mammals, mainly microtines (voles) (Carbyn 1995, Pattie and Fisher 1999, Moehrenschlager pers. comm.). Similarly, swift fox density in southeastern Colorado has been positively correlated with rodent abundance (Schauster *et al.* 2002). Elsewhere throughout its range, lagomorphs (rabbits and hares) appear to be important prey items for swift fox (Hines and Case 1991, Sovada *et al.* 2001).

An analysis of swift fox food habits in Alberta and Saskatchewan found that the mammalian component of their diet consisted primarily of small rodents (64.1%) and ungulate carrion (23.6%) (Reynolds *et al.* 1991). Lesser amounts of lagomorphs (5.2%) and ground squirrels were also consumed (Reynolds *et al.* 1991). Moehrenschlager *et al.* (2007a) found similar results for Canadian swift fox, with rodents as the most frequent prey item, found in 59.5% of scats, followed by insects (23.8%), birds (8.0%), lagomorphs (7.1%) and large mammals (1.7%). Moehrenschlager (2000) found that Richardson's ground squirrels comprised between 25% and 30% of swift fox diets south of the Cypress Hills, Saskatchewan. Richardson's ground squirrels are only consumed outside of the winter hibernation period, when they are active above-ground. Insects (in particular, grasshoppers) are also considered to be an important food source for swift foxes in Alberta and Saskatchewan, particularly in autumn (Moehrenschlager pers. comm.). Winter is considered to be the most critical food limiting period for swift foxes in Canada as that is when only microtines (voles), rabbits, carrion and some birds are available (Scott-Brown *et al.* 1986). Snow depth may also limit availability of food during the winter (Carbyn 1995).

In Nebraska and Texas, mammals and insects were the most frequently occurring items in swift fox scat (Hines and Case 1991, Kamler *et al*. 2007). Common mammalian species remains found in swift fox scat from Nebraska included: jackrabbit (*Lepus* spp.) (25%), prairie vole (*Microtus ochrogaster*) (48%), cattle (*Bos taurus*) (38%) and western harvest mouse (*Reithrodontomys megalotis*) (31%). Other food items of swift fox in Nebraska were plant material, found in 54% of scats, and birds, found in 40% of scats.

Swift fox are adaptable and are able to change their diets in response to the availability of different prey items in their habitat (Kamler *et al.* 2007). In Texas, swift fox inhabiting native prairie consumed more insects than swift fox in fragmented landscapes dominated by cropland and tame pasture. Swift fox inhabiting fragmented landscapes consumed more mammals, birds and crops than swift fox living in native prairie. Swift fox inhabiting fragmented landscapes had more diverse diets, probably reflecting the more diverse prey base contained in such landscapes (Kamler *et al.* 2007).

2.2.2 Predators

Several studies have found that predation by coyotes is the main cause of swift fox mortality throughout its range (Carbyn 1995, Sovada *et al.* 1998, Kitchen *et al.* 1999, Olson and Lindzey 2002a, Kamler *et al.* 2003, Ausband and Foreman 2007). Kamler *et al.* (2003) recently reported that coyotes were responsible for at least 89% of swift fox mortalities in the Rita Blanca National Grasslands in northwestern Texas. In Kansas, coyotes were responsible for 33% of swift fox deaths, which was the second leading cause of mortality behind non-traumatic injury (*e.g.*, starvation) (Matlack *et al.* 2000). In Montana, coyotes accounted for 46% of juvenile swift fox deaths (Ausband and Foresman 2007). The coyote is both a competitor and a predator of the swift fox (Brechtel *et al.* 1996, Kamler *et al.* 2003). Swift foxes killed by coyotes may be eaten, or killed and left (Brechtel *et al.* 1996). Coyote populations are much higher than in the past due to the elimination of wolves from the prairies and the ability of coyotes to adapt to modified prairie landscapes (Brechtel *et al.* 1996).

Other predators of swift fox include raptors and various mammal species. Predation by large raptors such as golden eagles (*Aquila chrysaetos*) is considered to have a significant impact on swift fox populations (Smeeton pers. comm.). Indeed, Ausband and Foresman (2007) found that 27% of juvenile mortalities from re-introduced swift foxes in Montana were from raptors, and Moehrenschlager *et al.* (2007a) found that golden eagles killed more swift foxes than coyotes at their study site. Predation by American badgers and bobcats (*Lynx rufus*) has also been recorded in the literature (Brechtel *et al.* 1996, Olson and Lindzey 2002a).

2.3 Habitat Requirements

2.3.1 General

Swift foxes generally prefer short, dry mixedgrass or mixedgrass prairie with flat to gently rolling terrain and sparse vegetation that allows for visibility and good mobility (Cotterill 1997, Kamler *et al.* 2003, Martin *et al.* 2007). Common native plant species found in dry mixedgrass and mixedgrass prairie include blue grama (*Bouteloua gracilis*), needle-and-thread (*Hesperostipa comata*), northern wheat grass (*Elymus* lanceolatus), western wheat grass (*Pascopyrum smithii*),

moss phlox (*Phlox hoodii*), scarlet mallow (*Sphaeralcea coccinea*) and pasture sage (*Artemisia frigida*). Swift fox typically avoid coulees and brushy areas with tall vegetation (Cotterill 1997).

Brechtel *et al.* (1996) identified three essential habitat features for swift fox:

- Extensive tracts of native grassland.
- Presence of burrowing mammals and soil types suitable for denning.
- Microtine rodent populations.

In Alberta, swift fox typically avoid cultivated areas (Cotterill 1997). Swift foxes also appear to avoid cultivated land in Texas (Schwalm *et al.* 2012). However, swift foxes in other parts of their range (*e.g.*, Kansas) will inhabit cultivated fields (Jackson and Choate 2000, Matlack *et al.* 2000, Sovada *et al.* 2009). Cropland may provide suitable habitat for swift fox provided that dryland practices are employed (Sovada *et al.* 2009). Fallow fields will occasionally be used by swift fox for denning and foraging (Sovada *et al.* 2009). Agricultural practices that use irrigation or that result in large monotypic crop fields will likely not be suitable for swift foxes (Sovada *et al.* 2009). Likely due to differences in farming practices, farmland in Canada does not appear to be as suitable for swift foxes as in parts of the United States (COSEWIC 2009).

As part of the Multiple Species at Risk (MULTISAR) conservation program in southern Alberta, a Habitat Suitability Index (HSI) model was developed for swift fox habitat use in the Milk River Watershed (Downey 2004a). Using information on known locations as well as the biology and ecology of wildlife species, HSI models use various physical and habitat variables to predict where suitable habitat for the species may be found on the landscape. According to the HSI model developed for the swift fox, the most suitable habitat for the species occurs in areas with flat terrain, silty soils, low shrub cover and high grass cover.

A second swift fox HSI model was developed independently of the MULTISAR work by the Centre for Conservation Research at the Calgary Zoo (Moehrenschlager *et al.* 2007b). This model was considered to be successful in predicting high quality habitat on the landscape and consisted of four variables: topography, cropland, moisture and habitat fragmentation. According to the model, swift fox prefer dry areas with gentle slopes away from cropland, roads, edge habitats and fragmented landscapes.

A Resource Selection Function (RSF) model was recently developed for swift fox habitat selection in southern Alberta, also as part of the MULTISAR program (Gan *et al.* 2014). RSFs are similar to HSI models but are thought to be more robust due to their rigorous and empirical evaluation of model performance using statistics (Boyce *et al.* 2002). RSFs estimate relative probability of occurrence for species of concern on the landscape based on various physical and anthropogenic variables. The best-fit RSF model as determined by Gan *et al.* (2014) contained five variables: percent shrub cover; proportion of ‘natural’ land; road density; average slope; and the proportion of favourable soil. According to the top model, relative probability of swift fox occurrence increases with lower shrub cover, lower road density, lower slopes, higher proportions of natural land, and in areas with suitable soils for denning (Gan *et al.* 2014).

2.3.2 Denning Habitat

Dens are a critical habitat component for swift fox. Swift fox use multiple den sites for shelter, rearing young and to escape from predators (Cotterill 1997). Availability of suitable dens significantly improves the odds of predator avoidance and is thought to be a critical factor affecting the maintenance of viable swift fox populations (Pruss 1999, Moehrenschlager pers. comm.). Most dens are built in natural substrates such as the burrows of fossorial mammals. However, anthropogenic structures such as rubble piles, culverts, pipes and buildings may also be used occasionally (Moehrenschlager *et al.* 2004). American badger burrows do not need to be further excavated and provide readily accessible predator escape dens. Swift foxes will also excavate their own dens or enlarge ground squirrel (*Spermophilus* spp.) or prairie dog (*Cynomys* spp.) burrows. Swift foxes are known to use dens throughout the year, and may even use the same one for their entire lifetime (Scott-Brown *et al.* 1986, Pattie and Fisher 1999).

Den location and physical characteristics are important criteria in determining the suitability of natal or rearing den sites (Pruss 1999). Swift fox will den in native prairie and cropland (Jackson and Choate 2000, Matlack *et al.* 2000). However, in Canada swift fox dens are almost exclusively in native prairie (Carbyn 1998). Natal or rearing dens typically have more entrances than temporary escape or shelter dens (Pruss 1999). Den locations vary depending on local topography; however, well-drained sites with good visibility in all directions are typically preferred (Cutter 1958b, Scott-Brown *et al.* 1986, Cotterill 1997, Jackson and Choate 2000). The tops of hills with a gradual slope provide favorable natal den sites (Pruss 1999). Locating dens on hilltops allows for improved predator detection and minimizes the probability that dens will be flooded during snowmelt or heavy rainfall (Pruss 1999). Dens have also been located in level terrain on shortgrass prairie (Cotterill 1997, Pruss 1999). Soil type and aspect may also be important criteria in den site selection (Hines and Case 1991). Hines and Case (1991) reported that 79% of 19 dens in Nebraska were located in sandy loam soils with the remainder in loamy sand. Dens with east and west exposures were also preferentially selected (Hines and Case 1991). No preference was found for a particular den entrance angle for dens in southeastern Alberta and southwestern Saskatchewan (Pruss 1999).

In Texas, it was observed that swift foxes constructed dens in “overgrazed”, barren areas with little or no bushes or tall plants (Cutter 1958b). However, in southeastern Alberta and southwestern Saskatchewan, Pruss (1999) found that occupied natal dens were located in areas where old grass was significantly higher than at unoccupied sites. Pruss (1999) suggests that the preference for relatively taller grass may be related to the availability of insect prey. Similarly, Uresk *et al.* (2003) found that vegetation at swift fox dens in South Dakota had a greater vegetation height density than random locations (approximately 12 cm versus 10 cm visual obstruction readings [VOR]). Total vegetation height significantly improves hiding and screening cover for foxes at dens (Uresk *et al.* 2003).

Both Pruss (1999) and Hines and Case (1991) found an association between proximity of dens to roads. Hines and Case (1991) reported that 68% of swift fox dens were within 230 m of roads; similarly, Pruss (1999) found that 81% of occupied dens were within 200 m of roads, and that on average dens were situated 267 m from the nearest road and 1 km away from the nearest water source. Pruss (1999) speculates that proximity to roads may be due to avoidance of coyotes,

availability of vehicle-killed carrion as a food source and the use of roads as travel corridors. However, the proximity of dens to roads may also increase potential vehicle collision mortalities. Black *et al.* (1998) found that vehicle trauma was a significant cause of death among young-of-the-year swift foxes in Alberta and Saskatchewan.

2.3.3 Foraging Habitat

Selection of suitable foraging sites is influenced by various factors, including avoidance of predators or competitors such as coyotes and red foxes, availability of escape dens and cover and availability of suitable prey (Moehrenschlager pers. comm.). In general, swift fox forage in open mixedgrass prairie with short vegetation that allows for ease of movement and foraging efficiency (Smeeton pers. comm.). Swift fox are more vulnerable to predators in taller vegetation and therefore select for open vegetation with greater visibility to avoid predation by larger carnivores such as coyotes (Thompson and Gese 2007, Moehrenschlager pers. comm.). Cattle grazing is considered important for maintaining patches of suitably short vegetation (Smeeton pers. comm.). Although there is usually an abundance of various prey available to swift foxes during the summer, availability of mammalian prey during the winter is considered limiting (Carbyn 1995). Vegetation patchiness and structural diversity is considered important for maximizing the availability and stability of a diverse mammal prey base (Fagerstone and Ramey 1996).

Uresk *et al.* (2003) found that the vegetation characteristics of swift fox foraging habitat in South Dakota were similar to denning habitats, with more dense vegetation than random sites. In a study of habitat use by female swift fox during the pup-rearing season, Sasmal *et al*. (2011) observed that female swift fox in South Dakota used grassland, sparse vegetation and prairie dog towns in proportion to their availability. In contrast, female swift foxes used woodland, shrubland, pasture/agricultural land and developed areas in lower proportion relative to their availability.

Olson and Lindzey (2002a) attribute the higher productivity of swift fox in their southeastern Wyoming study area to a positive association between the proportion of sage vegetation within the parent's home range and prey abundance in sagebrush habitats. Olson and Lindzey (2002a) noted that sagebrush-grassland habitats had higher relative abundance of prey (including small mammals, breeding birds and insects) than grassland habitats in southeastern Wyoming. Important prey items such as thirteen-lined ground squirrels (*Spermophilus tridecemlineatus*), deer mice (*Peromyscus maniculatus*) and various beetles have all been positively associated with shrub microhabitats (Olson and Lindzey 2002a). Additional research is needed to define the characteristics and seasonal variation of swift fox foraging habitat and microhabitat selection in Alberta and Saskatchewan.

2.3.4 Area Requirements

Swift fox home range sizes exhibit spatial and temporal variation, and also appear to be related to prey availability and distribution as well as habitat productivity and physical attributes (Hines and Case 1991, Olson and Lindzey 2002b, Schauster *et al.* 2002). Based on data from 16 swift foxes in the Alberta/Saskatchewan border area, Cotterill (1997) reported an average home range size of approximately 34.1 km^2. Similarly, using location data for 36 individuals in the Canada-

United States border region, Moehrenschlager *et al.* (2007a) estimated swift foxes to have an average home range size of 31.9 ± 4.8 km^2. This is similar to an average home range estimate of 32.3 km^2 reported for adult swift foxes in Nebraska (Hines and Case 1991). However, home ranges reported for swift foxes on native shortgrass prairie in southeastern Colorado and in southeastern Wyoming were significantly smaller (Olson and Lindzey 2002b, Schauster *et al*. 2002). For example, average annual home range size of swift foxes in southeastern Wyoming was approximately 19 km^2 (Olson and Lindzey 2002b). Thompson *et al*. (2008) and Thompson and Gese (2012) reported average home range sizes of 6.53 km^2 and 15.4 km^2 in southeastern Colorado. Lebsock *et al.* (20120 observed even smaller annual average home range sizes (4.18 km^2) in northeastern Colorado. The same authors reported an annual core area size (*i.e.*, the area where it spends 50% of its time) as 0.53 km^2.

Schauster *et al.* (2002) found that swift fox home ranges in southeastern Colorado were largest at night (9.4 ± 4.9 km^2). Spatial overlap of swift fox home ranges was also greatest at night. During the day, swift fox social units, ranging from two to four foxes, used dens and had reduced home ranges of 2.8 ± 2.2 km^2. Denning areas were not commonly shared between social units. These findings suggest that while nocturnal foraging areas are commonly shared by various social groups, swift foxes showed some degree of territoriality toward denning areas (Schauster *et al.* 2002). Olson and Lindzey (2002b) also suggest that swift foxes may be territorial due to limited core-area home range overlap between adjacent pairs. Moehrenschlager *et al.* (2007a) reported that swift fox home ranges overlapped 74% among mates and only 29% between neighbours. Swift fox home ranges have been noted to expand in the winter in order to accommodate reduced prey availability (Cotterill 1997, Olson and Lindzey 2002b). The home ranges of both males and females also appears to decrease in size during the pup-rearing period, with the change greater for females (Olson and Lindzey 2002b). Similarly, Lebsock *et al*. (2012) found that swift fox home ranges are larger in the breeding season compared to both the pup-rearing and dispersal seasons; home range sizes were not significantly different between the pup-rearing and dispersal seasons. These same authors also found that while male home ranges tended to be larger than that of females, the difference was not significant.

Swift fox densities have been found to increase in better quality habitats and have been positively correlated with rodent prey abundance (Schauster *et al.* 2002). Swift fox density may be inversely related to home range size as home-range dispersion patterns may be influenced by the presence of other fox pairs (Olson and Lindzey 2002b). Olson and Lindzey (2002b) suggest that home range size may be an indicator of habitat productivity, with decreasing home range size in more productive habitats. Home range variations may also be due to natural physical boundaries, such as roads or rivers, which constrict home range boundaries (Hines and Case 1991).

Similar to home range sizes, dispersal distances of swift foxes leaving their natal dens also appear to vary geographically. In southeastern Colorado, Covell (1992) reported dispersal distances of 9.4 ± 1.7 km for juvenile males and 2.1 ± 0.2 km for juvenile females, whereas Schauster *et al.* (2002) reported that juveniles dispersed distances ranging from 8.4 km to 15.9 km. Ausband and Foresman (2007) reported that dispersal distances for juvenile swift fox in northern Montana averaged 10.4 km, with no significant difference between males and females. For this same Montana population, Ausband and Moehrenschlager (2009) recorded a number of

juvenile swift foxes dispersing more than 50 km from their natal dens, with one female travelling more than 190 km, eventually joining a Canadian population.

3 AMERICAN BADGER

3.1 Background

American badgers are medium-sized carnivores with bold facial markings and flattened torsos, the latter an adaptation to living underground. The species is considered a top predator in the prairie ecosystem. The American badger is a member of the Weasel family (Mustelidae) and the only species of badger occurring in North America. Of the four subspecies of American badger that are found in North America, three live in Canada. *Taxidea taxus taxus* is the subspecies that occurs in Alberta. It is found in the Parkland and Grassland Natural Regions from the North Saskatchewan River in the north to the Montana border in the south and from the foothills in the west to the Saskatchewan border in the east (Scobie 2002, NRC 2006). *Taxidea taxus taxus* occurs throughout the southern half of the Prairie Provinces (COSEWIC 2012). The extent of occurrence of this subspecies in Canada is 721,096 km^2 (COSEWIC 2012). The other two badger subspecies, *Taxidea taxus jeffersonii* and *Taxidea taxus jacksoni*, occur in British Columbia and southern Ontario, respectively (COSEWIC 2012).

The American badger is currently listed as 'Sensitive' under the Alberta General Status listing (Government of Alberta [GoA] 2012). The species is listed as 'Data Deficient' under the provincial *Wildlife Act*. It is also considered a 'fur-bearing animal' under the *Act*, meaning it is allowed to be hunted in Alberta with a proper license. Nationally, the American badger, including all four subspecies, was originally designated as 'Not at Risk' by COSEWIC in 1979 (Stardom 1979). The four subspecies were separated out and given individual status designation in 2000, with *Taxidea taxus taxus* retaining the 'Not at Risk' listing (COSEWIC 2000b). In 2012, the status of *Taxidea taxus taxus* was re-examined and the species was designated as a species of 'Special Concern' (COSEWIC 2012). Currently, *Taxidea taxus taxus* remains a species of 'Special Concern' in Canada (COSEWIC 2016). Internationally, the IUCN has listed the American badger as a species of 'Least Concern' (Reid and Helgen 2008).

No reliable population estimates exist for American badgers in Alberta (COSEWIC 2012). The last attempt at a provincial American badger population estimate was by COSEWIC (2000), which estimated there were 1,000 to 10,000 animals in the province, based on a survey of Alberta wildlife managers. Historically, the American badger population in Canada was significantly impacted by trapping harvests (Scobie 2002). Currently, even the most generous American badger population estimates are only about half of the known 18,000 badger pelts that were harvested in 1928 (Scobie 2002). Between 1999 and 2009 an average of 231 American badger pelts were sold annually in Alberta (COSEWIC 2012). Although more information is needed regarding American badger population trends in Alberta, populations are thought to be declining provincially (Scobie 2002).

Although American badgers have expanded their range in some areas due to the clearing of forests, cultivation is thought to have decreased suitable badger habitat on the prairies (Scobie

2002). Indeed, habitat loss and alteration together with trapping harvest, road kills, shooting, poisoning and suspected declines of essential prey species (*i.e.*, northern pocket gopher [*Thomomys talipoides*] and Richardson's ground squirrel) are considered to be limiting factors for American badger populations in Alberta (Scobie 2002). American badgers are considered ecologically important in the prairie landscape because their burrows provide cover, nesting and denning opportunities for other wildlife species, including swift fox, snakes, invertebrates, burrowing owls (*Athene cunicularia*) and a host of other organisms (Carbyn *et al.* 1999, Scobie 2002). American badger diggings also help to aerate the soil, promote the formation of humus and allow water to quickly reach deeper soil levels (Scobie 2002).

3.2 Ecology

American badgers are nocturnal, fossorial (digging, burrowing) mammals. American badgers typically mate in July and August, and have delayed implantation until February resulting in a spring birth (Pattie and Fisher 1999, Scobie 2002). American badgers are polygynous, with males having more than one mate. Between April and June, one to five young are born in a litter. At three to four months of age, the young disperse and are sometimes seen in unsuitable habitat (Scobie 2002). Most American badgers reach sexual maturity by one year of age (Pattie and Fisher 1999). American badgers can live up to 14 years of age; however, most do not live more than four years (Scobie 2002). American badgers enter into torpor (a period of reduced activity) in their burrows during the winter (Scobie 2002). American badgers catch most of their fossorial and semi-fossorial prey by digging them out of the ground (Michener 2004). American badgers hunt primarily at night and remain underground during the day (Scobie 2002).

American badgers excavate large volumes of soil during their foraging activities. Eldridge (2004) estimated the species moves up to 26 tonnes/ha of soil. American badger diggings, scratchings and mound excavations influence a range of soil physical processes, including improving water infiltration, and aeration, relieving compaction and porosity (Hole 1981, Laundré and Reynolds 1993). Mounds, their holes and associated scratchings also tend to trap essential resources, such as organic matter, soil, litter and water, which likely lead to enhanced levels of nitrogen and carbon in holes (Eldridge 2004). Burrows created by American badgers are often used by other wildlife species in Alberta, including burrowing owls (*Athene cunicularia*) and swift fox, among others (Cotterill 1997, Poulin *et al.* 2005).

3.2.1 Diet

American badgers are mainly carnivorous, opportunistic feeders and feed primarily on northern pocket gophers and Richardson's ground squirrels in Alberta (Sovada *et al.* 1999, Scobie 2002). Time of year and American badger maturity influence the availability and selection of food (Sovada *et al.* 1999). Michener (2000) found that American badgers in southern Alberta cache and retrieve Richardson's ground squirrels. It was suggested that American badgers cache food for future energetically stressful periods. The two main high energy demand periods occur during winter and lactation. The caching of ground squirrels was concentrated in autumn, during October and November, with cache retrieval extending into early December (Michener 2000). Obtaining excess food reserves in autumn is important for American badgers to prepare for the winter and prolonged periods of reduced activity. American badger predation in autumn

coincided with periods when most ground squirrels were in hibernation. A concentrated period of American badger predation on un-weaned infant ground squirrels in the spring was also reported, prior to their emergence above ground (Michener 2000).

A seasonal prey and food study conducted in central Alberta, revealed that northern pocket gophers were considered the major food item from late March or early April to early July (Salt 1976). Richardson's ground squirrels made up a dominant component of the American badger diet from mid-July to early October. In October, meadow voles (*Microtus pennsylvanicus*) and insects, especially beetles and grasshoppers, accounted for the majority of their diet (Salt 1976). Birds, eggs, reptiles, amphibians, invertebrates, fish, mollusks and plant material are commonly used to supplement the American badger's diet (Scobie 2002).

Elsewhere, American badgers in British Columbia fed primarily on Columbian ground squirrels (*Spermophilus columbianus*), voles, insects, amphibians and other rodents (*e.g.*, northern pocket gopher) (Kinley and Newhouse 2014). In Montana, American badgers prey on black-tailed prairie dogs (*Cynomys ludovicianus*) (Eads and Biggins 2008). American badgers in Colorado will prey on yellow-bellied marmots (*Marmota flaviventris*) (Armitage 2004). Duck eggs were found to be commonly consumed by American badgers in Minnesota and North Dakota during the duck nesting season (Sovada *et al.* 1999). American badgers in Ontario will prey on eastern cottontails (*Sylvilagus floridanus*) and woodchucks (*Marmota monax*) (Dobbyn 1994). American badgers have also been known to scavenge carrion such as cattle, lagomorphs and road-kill (Sovada *et al.* 1999).

3.2.2 Predators

Common American badger predators include bears *(Ursus* spp.*)*, other badgers, wolves (*Canis lupus*), cougars (*Felis concolor*), bobcats (*Lynx rufus*), golden eagles (*Aquila chrysaetos*) and bald eagles (*Haliaeetus leucocephalus*) (Scobie 2002, Kinley and Newhouse 2008). Predation by coyote packs on American badgers has also been reported (Scobie 2002).

3.3 Habitat Requirements

3.3.1 General

American badgers are commonly found in open grasslands with friable soils, little woody cover (shrubs or trees) and within close proximity to a suitable food source, such as Richardson's ground squirrel colonies (Scobie 2002). Abundance of prey is considered a strong factor in determining the distribution of American badgers (Apps *et al.* 2002). American badger occurrence has also been shown to decrease with increased habitat fragmentation and decreasing habitat patch size (Crooks 2002).

Similar to the swift fox, an HSI model was developed for the American badger as part of the MULTISAR conservation program. The model for American badgers in the Milk River Watershed included four variables: soil texture; graminoid coverage; slope; and roadways (Downey 2004a). According to the HSI model, quarter-sections which contained steep slopes, less than 70% graminoids, coarse or very fine soils and that were close to main roadways were rated as poor American badger habitat. Moderately coarse (sandy loam), medium (loam, silt

loam and silt), and moderately fine (sandy clay loam, silty clay loam, and clay loam) soils are considered preferable for burrow construction and stability. Graminoid coverage of 20% was chosen as the minimum requirement for suitable American badger habitat, while habitats with greater than 70% graminoids were given an optimal suitability index rating. Slopes between 0° and 15° were determined to be most representative of suitable American badger sites. A buffer of 400 m to the nearest roadway was considered optimal to reduce the risk of vehicle mortalities.

As an update to the MULTISAR American badger HSI model, an RSF model was developed for the badger in 2011 (Skilnick and Dodd 2011). The best-fit model, which performed better than the original HSI model, contained seven variables: distance to glaciofluvial and glaciolacustrine deposits; distance to Regosolic soils; soil drainage; percent graminoid cover; road density; and elevation. According to the top model, American badger relative probability of use increases: at lower elevations; closer to glaciolacustrine deposits; further from glaciofluvial deposits; further from Regosolic soils; with lower road densities; with higher graminoid cover; and with presence of well-drained soils.

American badgers in southeastern British Columbia were most often associated with dry, open range and southern aspects (Apps *et al.* 2002). Preferred American badger habitat was also positively associated with linear disturbances, fine sandy-loam textured soils, open habitat and glaciofluvial deposits (Apps *et al.* 2002). A negative association was found with gravelly areas, forest cover, elevation, ruggedness and colluvial deposits. Kinley *et al*. (2013) conducted RSF modelling of American badger habitat use in southeastern British Columbia. Their top-ranked model included elevation, slope, solar radiation, ecoclass (expected climax vegetation), crown closure, leading tree species, soil order and parent material as well as elevation:slope and elevation:solar radiation interactions. American badgers in this area were selected for low elevations, shallow slopes, high solar radiation, low crown closure and Brunisolic soils (Kinley *et al*. 2013).

Duquette *et al.* (2014) conducted RSF modelling for American badger habitat use in a highly fragmented agricultural landscape in Ohio and Illinois. Their results showed that American badgers in Illinois selected for native prairie, cropland, higher elevation sites and, to a lesser extent, pasture and upland forest, while avoiding permanent water and roads. American badgers in Ohio selected for pasture, upland forest, roads and, to a lesser extent, native prairie and high elevation sites, while avoiding permanent water.

3.3.2 Denning Habitat

A burrow or den is vital to an American badger as it is used for giving birth and rearing young, food storage, foraging and as a diurnal activity site throughout the year (Scobie 2002). Dens may be reused by the same or other American badgers and are variable in characteristics (Lindzey 1976). Soil texture is an important factor (see HSI model). American badgers dig their own dens or excavate Richardson’s ground squirrel burrows (Pattie and Fisher 1999). A natal den may be up to 10 m long and have a diameter of 30 cm (Scobie 2002). Natal dens typically only have one entrance and a mound twice the size of day-use dens (Lindzey 1976). A nest consists of bulky grasses in an expanded chamber. Sargeant and Warner (1972), found that radio collared American badgers were never located in more than one den on any day. In fact, a female

American badger was noted to have only reused two of her 46 known dens and was never found in the same den on two consecutive days in a summer season (Sargent and Warner 1972). When observed in the winter, the winter den was the last used den in the summer activity period.

3.3.3 Foraging Habitat

As American badgers are opportunistic foragers, they alter their diet and change the location of their preferred feeding sites in response to prey availability (Salt 1976). Suitable American badger foraging habitat in southern Alberta typically corresponds with areas of high concentrations of Richardson's ground squirrels (Michener 2000, Scobie 2002). The habitat requirements of Richardson's ground squirrels are described in Section 4.3, below.

3.3.4 Area Requirements

The American badger's home range size in Alberta is unknown. However, studies from the United States indicate that American badger home range sizes are variable and change seasonally and by sex (Sargeant and Warner 1972, Lindzey 1978, Scobie 2002). Females tend to be spatially isolated and have smaller home ranges than males (Lindzey 1978, Scobie 2002, Duquette *et al.* 2014). Male home ranges typically overlap and may triple in size during the breeding season to maximize access to fertile females (Goodrich and Buskirk 1998, Scobie 2002). In Idaho, home ranges in autumn and winter (Lindzey 1978) were reportedly larger than home ranges during the spring and summer (Messick and Hornocker 1981). Lindzey (1978) reported average home ranges of 583 ha for males and 237 ha for females during autumn and winter. Messick and Hornocker (1981) reported average home ranges of 170 ha for males and 130 ha for females during the spring and summer. In contrast, home range size for a female American badger in Minnesota increased greatly from 2 ha in winter to 761 ha in summer, and then decreased to 53 ha in autumn (Sargeant and Warner 1972). Duquette *et al.* (2014) reported average home range sizes of 5,672 ha and 1,320 ha for male and female American badgers, respectively, in Illinois and 240 ha and 961 ha for males and females, respectively, in Ohio.

American badgers in British Columbia (*Taxidea taxus jeffersonii*) appear to have larger home range sizes than reported for the United States (Kinley and Newhouse 2008, Hoodicoff *et al.* 2009). Hoodicoff *et al.* (2009) found that home ranges averaged 7,860 ha, with a range of 900 ha to 25,840 ha. Similarly, Kinley and Newhouse (2008) observed average home range sizes of 30,100 ha and 3,500 ha for males and females, respectively. Some American badgers in British Columbia also appeared to have core areas within their home ranges where they spent the majority of their time (Hoodicoff *et al*. 2009).

Resident and disperser types of American badgers have been identified (Carbyn *et al.* 1999). Dispersers may travel long distances from their natal den and move erratically, whereas residents have established territories that can change in size seasonally depending on food availability. The longest reported dispersal distance of an American badger was 52 km for a female and 110 km for a male (Messick and Hornocker 1981). Hoodicoff *et al.* (2009) also found that American badgers in British Columbia were capable of travelling long distances in short periods of time, with average 12 hour movement rates of 645 m in summer and 48 m in winter. The greatest movement observed by Hoodicoff *et al.* (2009) was of a male that travelled over 14 km in a 4 hour period at night.

4 RICHARDSON'S GROUND SQUIRREL

4.1 Background

The Richardson's ground squirrel is a small, ground-dwelling mammal of the order Rodentia. It occurs in the Prairie Provinces in Canada and in that states of Montana, South Dakota, North Dakota and western Minnesota in the United States (Michener 2002). In Alberta, Richardson's ground squirrels are common and widespread throughout the Dry Mixedgrass, Mixedgrass, Northern Fescue and Foothills Fescue Natural Subregions (Michener and Schmutz 2002, NRC 2006). Richardson's ground squirrel are a critical part of the prairie ecosystem as they are a common prey species of many wildlife species and their burrows provide critical underground refuges for numerous bird, mammal, reptile and insect species (Michener and Schmutz 2002). As discussed previously, Richardson's ground squirrels are among the primary prey consumed by American badgers and are also an important prey item for swift fox. Both American badgers and swift fox will also excavate and use Richardson's ground squirrels burrows as dens.

Richardson's ground squirrels are currently listed as 'Secure' in Alberta under the General Status listing (GoA 2016). The species is considered a 'non-license animal' under the provincial *Wildlife Act*, meaning they can be hunted without a license. The Richardson's ground squirrel has not been assessed nationally by COSEWIC. Internationally, the IUCN lists Richardson's ground squirrels as a species of 'Least Concern' (Lindzey and Hammerson 2008).

Limited monitoring has been done to assess Richardson's ground squirrel population numbers or long-term trends at a local or regional scale in Alberta (Michener and Schmutz 2002). However, recent outbreaks in numbers of Richardson's ground squirrels in the Prairie Provinces suggest that populations remain healthy (Proulx 2010). Proulx (2010) attributed the high densities of Richardson's ground squirrels in recent years to several factors, including drought, poor grassland management, low cattle prices, inefficient rodenticides, loss of predators and loss of family farms.

Despite their ecological value, ground squirrels are considered 'pests' and are routinely poisoned as part of rodent control programs on agricultural lands. Road-kills are another significant cause of human-induced mortality (Gadd 1995). Although ground squirrels have adapted somewhat to human-modified habitats, much of their natural, native prairie habitat has been fragmented or lost to cultivation and development (Michener 2002).

4.2 Ecology

Richardson's ground squirrels are diurnal (active during the day), semi-fossorial mammals that spend the majority of their life underground. Richardson's ground squirrels live in clusters of female kin with each female having her own burrow system and rearing her own litter by herself (Saskatchewan Environment and Resource Management [SERM] 2001). In southern Alberta, adult male Richardson's ground squirrels usually emerge from hibernation in late February to early March, while adult females emerge about two weeks later (Michener 2002). The active season can begin two to three weeks later in more northerly locations in Alberta and can also vary between years depending on winter severity. Mating begins in the spring after female Richardson's ground squirrels emerge from hibernation (Michener and Schmutz 2002). Females

will mate with multiple males, resulting in litters with varying paternities (Hare *et al.* 2004). The gestation period is approximately 23 days (Michener and Schmutz 2002). All litters are usually born within a one to three week period, and all young squirrels typically emerge at the same time (Michener 2002). Juveniles will appear above ground in early to mid-May when they are 28 to 30 days old. Litters are usually comprised of seven to eight young, but may range from three to 11 young (Pattie and Fisher 1999). Young are weaned at one month of age and their growth is rapid throughout the summer. Both males and females are sexually mature at one year of age (Michener and Schmutz 2002). Natural mortality is a major factor in regulating population density as less than 1% of males reach three years of age and only 40% to 50% of juvenile females survive to adulthood (Michener and Schmutz 2002).

Interestingly, Richardson's ground squirrels spend the majority of their time in hibernation and are only above ground for 15% of their lifetime (Michener and Schmutz 2002). Typically, adult Richardson's grounds squirrels will hibernate for four to eight months. Adult males typically enter hibernation around mid-June to early July, while females go into hibernation in early to late July (Michener 2002). Juveniles remain active until early to mid-August (females) or mid-September to October (males). During hibernation, Richardson's ground squirrels rely on stored fat to accommodate their metabolic demands (Michener 2000).

Richardson's ground squirrels are safe from most predators during hibernation with the exception of American badgers; badgers are the only species that can gain access to ground squirrels within their dens while they are hibernating (Michener 2000, 2002). As hibernating Richardson's ground squirrels are in a state of torpor and also because of the lack of escape routes in their hibernation system, they make easy prey. During hibernation as many as 50% of hibernating Richardson's ground squirrels can be killed by American badgers (Michener 2002).

4.2.1 Diet

Richardson's ground squirrels are primarily herbivorous, with vegetation (including roots, leaves, flowers and seeds) constituting between 80% and 100% of their total diet (Michener and Schmutz 2002). Richardson's ground squirrels eat native vegetation and the seeds and seedlings of forage and cereal crops, such as common wheat (*Triticum aestivum*), cultivated barley (*Hordeum vulgare*) and cultivated oat (*Avena sativa*) (Michener and Schmutz 2002). Richardson's ground squirrels also consume insects and are known to opportunistically scavenge carrion, such as road-killed small mammals (Michener and Schmutz 2002).

4.2.2 Predators

Richardson's ground squirrels are a vital prey source for various wildlife species, including swift fox, American badger, coyotes, long-tailed weasels (*Mustela frenata*), ferruginous hawks (*Buteo regalis*), Swainson's hawks (*Buteo swainsoni*), red-tailed hawks (*Buteo jamaicensis*), prairie falcons (*Falco mexicanus*), great horned owl (*Bubo virginianus*) and prairie rattlesnakes (*Crotalus viridis*) (Michener 2000, Michener 2001, Moehrenschlager 2000, Scobie 2002).

4.3 Habitat Requirements

4.3.1 General

Richardson's ground squirrel colonies typically occur in open, flat, dry, upland grasslands with sufficiently short vegetation to allow them to detect predators from a distance (Michener and Schmutz 2002). Croplands and tall vegetation is considered unsuitable habitat; however, Richardson's ground squirrels do occur along the edges of cultivation and in ditches (Michener 2002). Richardson's ground squirrels are found in greater numbers in flat, heavily grazed areas than in taller vegetation (Michener and Schmutz 2002). Downey *et al.* (2006) found that Richardson's ground squirrels selected for native prairie and against cropland. Within native prairie, Richardson's ground squirrels preferred vegetation ≤30 cm in height (Downey *et al.* 2006). Proulx and MacKenzie (2009) suggested that, within agricultural areas (*i.e.*, cropland, hay land and tame pasture), Richardson's ground squirrels preferred fields with vegetation heights ≤15 cm.

Results from a survey along thirty, 8 mile (12.9 km) transects randomly distributed in the Grassland Natural Region of Alberta (NRC 2006) showed that significantly more Richardson's ground squirrels were found in native prairie than in modified habitats (Downey 2003b, unpublished survey results). Out of a total of 796 Richardson's ground squirrels recorded during this survey, 571 (72%) were found on native prairie (n=696); 128 (16%) were found in cultivation (n=808); 58 (7%) in tame pasture (n=141); 32 (4%) in irrigated hay land (n=15); and 7 (0.9 %) in farmyards (n=24). The majority of Richardson's ground squirrels found in native prairie were in short vegetation (<10 cm). Schmutz (1989) observed lower densities of Richardson's ground squirrels in areas with greater than 30% cultivation compared to areas with less than 30% cultivation.

MULTISAR developed an HSI model for Richardson's ground squirrels in the Milk River Watershed in the early 2000s (Downey 2004c). The model included three, equally-weighted variables: grassland coverage, slope and soil texture. Quarter-sections with at least 20% graminoid cover, slopes ≤10° and soils with moderately coarse (sandy loam) or medium (loam, silt loam or silt) textures were rated as optimum Richardson's ground squirrel habitat.

4.3.2 Denning Habitat

Laundré and Appel (1986) found that Richardson's ground squirrel burrow sites in southwestern Minnesota were preferentially located in short vegetation (≤5 cm) and moderately well-drained soils. Short vegetation is thought to be important not only for predator detection but also for social communication involving visual cues (Laundré and Appel 1986, Michener and Schmutz 2002). Suitable soils with sufficient drainage are important to prevent burrows from flooding, to ensure ease of digging and to maintain the structural integrity of burrows (Laundré and Appel 1986, Michener and Schmutz 2002). In Alberta, Richardson's ground squirrels tend to prefer sandy and well-drained soils and typically avoid loose sand or heavy clay soils (Reynolds *et al.* 1999).

Burrows of Richardson's ground squirrels are quite intricate with several secondary entrances and "plunge" holes (Pattie and Fisher 1999). "Plunge" holes are inconspicuous entrances that are

used for quick escapes (Gadd 1995). A main burrow chamber may be 4 m to 10 m in length and usually ends in a grass-lined nest chamber (Michener 2002). Burrows can extend up to 1.8 m in depth (SERM 2001).

4.3.3 Foraging Habitat

Richardson's ground squirrels typically forage near to their burrows to ensure quick escape from predators. Therefore, sites that are suitable for burrowing, as described above, are also used for foraging. As Richardson's ground squirrels have a broad diet and consume a variety of native and non-native plants, short vegetation height is considered more important than species composition in determining suitable foraging habitat (Michener and Schmutz 2002). However, as noted by Michener and Schmutz (2002), more information is needed about the natural diet and foraging preferences of Richardson's grounds squirrels in Alberta.

4.3.4 Area Requirements

Home ranges for female Richardson's ground squirrels may be up to 240 m^2 in size in the summer. However, they tend to spend the majority of their time in core areas of 20 m^2 to 40 m^2 (Michener 2002). Female relatives may have overlapping home ranges (Michener 2002). Female home range sizes tend to increase until early summer and subsequently decrease closer to the hibernation period (Michener 2002). Male home ranges also fluctuate and typically increase up to mating and then decrease substantially (Michener 2002). During the breeding season, male home ranges can overlap the areas used by as many as 10 females (Michener 2002).

5 GRAZING AND BURROWING MAMMALS

There have been few studies that have assessed how cattle grazing or selective grazing systems affect swift fox, American badgers, or Richardson's ground squirrels in Alberta or across their range in North America. However, based on their ecological requirements, grazing appears to be necessary for maintaining suitable habitat for these species or their prey.

5.1 Swift Fox Response to Grazing

Open, unobstructed areas with short vegetation are considered important to swift foxes to enable them to catch prey efficiently and avoid predation (Smeeton pers. comm.). Swift foxes are built for speed, and in open areas can reach speeds of over 50 kilometers per hour (FWD 1990). Given their small size, taller vegetation impedes their natural running stride (Smeeton pers. comm.). Therefore, grazing is considered important for maintaining suitable habitat for this species (Smeeton pers. comm.). However, the benefits of grazing to swift fox depend in particular on the effects of grazing on winter prey availability (Carbyn 1989, Moehrenschlager pers. comm.). Although food is not considered limiting to swift fox during the summer months, availability of small mammal prey during the winter is critical to their survival (Carbyn 1989). Heavy grazing has been shown to negatively affect rodent species diversity in arid regions and has been correlated with declines in the kit fox (*Vulpes macrotis*) in the United States (Rosenzweig and Winakur 1969, Brechtel *et al.* 1996). Heavy grazing is thought to be particularly detrimental to most vole species which tend to prefer areas with sufficient canopy cover (Fagerstone and Ramey 1996). Sufficient vegetation cover is particularly important to voles during winters with

limited snow cover (Moehrenschlager pers. comm.). Therefore, heavy grazing over large areas may be detrimental to swift fox if it reduces prey availability during the winter months (Brechtel *et al.* 1996, Moehrenschlager pers. comm.). Since swift fox are nocturnal, heavy grazing systems that favour diurnal prey species may also be detrimental to foxes as hungry foxes would hunt whenever prey is available, increasing their vulnerability to predation (Carbyn 1989).

Heavy grazing over large areas may not only be detrimental in terms of reducing winter prey availability, but might also negatively affect availability of preferred swift fox breeding sites. Recent studies in Alberta, Saskatchewan and South Dakota indicate that swift fox breeding dens are preferentially placed in areas with taller grass cover, potentially due to improved hiding and screening cover (Pruss 1999, Uresk *et al.* 2003). A patchy environment with areas of variable degrees of grazing intensity is therefore considered more appropriate for maintaining swift fox habitat (Carbyn 1989). Smaller, heavily grazed patches offer suitable swift fox foraging habitat, while light to moderately grazed patches provide denning cover and refuge habitat for their small mammal prey. This type of vegetation patchiness was thought to have been promoted by former bison grazing regimes (Carbyn 1989).

5.2 American Badger Response to Grazing

Recent research in Grassland National Park in Saskatchewan is starting to shed light on the effects of livestock grazing on various wildlife species, including American badgers and ground squirrels (Bylo *et al.* 2014). This research examined the effects of different livestock grazing intensities on abundances of ground squirrel and American badger burrows in upland and lowland habitats in the park. Findings from this study suggest that in upland habitats, American badger burrows increase in abundance as grazing intensity decreases and as litter cover and vegetation biomass increase. In contrast, abundance of ground squirrel burrows in upland habitats increased as grazing intensity increased and as vegetation biomass decreased. Abundance and occurrence of both ground squirrel and American badger burrows in lowland habitats was relatively independent of grazing intensity or vegetation. Because of the apparent contrasting effects of livestock grazing on these two mammal species, Bylo *et al.* (2014) suggest that a range of grazing intensities may be needed to maintain high diversity of burrowing mammals in prairie environments.

It has been suggested that in contrast to Richardson's ground squirrels, northern pocket gophers, the other main prey species of American badgers, are attracted to Healthy rangeland with vigorous plant growth, high vegetation biomass with abundant forbs and even ungrazed areas (Fagerstone and Ramey 1996). However, in Alberta high numbers of northern pocket gophers are often associated with modified and heavily grazed rangeland (Adams *et al.* 2009). Modified and heavily grazed sites contain weedy, often tap-rooted species (*e.g.*, common dandelion [*Taraxacum officinale*]) preferred by pocket gophers (Adams *et al.* 2009). Because American badgers are generally found in lightly grazed areas while their prey are found in heavily grazed areas (Bylo *et al.* 2014), grazing systems that promote vegetation patchiness will likely be of benefit to the badger. Appropriate vegetation patchiness can be achieved with continuous grazing at light to moderate grazing intensity or with rotational grazing where the fields are large enough (or grazing intensity light enough) to allow livestock to forage selectively. Grazing systems that

promote vegetation patchiness will also encourage habitat for a diversity of other suitable small mammal prey (Fagerstone and Ramey 1996).

To refine grazing recommendations in relation to American badgers, studies need to define the vegetation structure preferred by badgers at natal (breeding) dens. Furthermore, studies should evaluate threshold densities of Richardson's ground squirrels and pocket gophers that are needed to maintain a viable population of American badgers.

5.3 Richardson's Ground Squirrel Response to Grazing

Moderate to heavy grazing intensities are considered beneficial to Richardson's ground squirrels, which rarely colonize areas with tall, dense vegetation (Fagerstone and Ramey 1996, Bylo *et al.* 2014, Michener pers. comm.). As previously discussed, Richardson's ground squirrels in Grasslands National Park responded positively to increased grazing intensity and reduced vegetation biomass in upland habitats (Bylo *et al.* 2014). Richardson's ground squirrels require shorter vegetation (≤10 cm) to easily detect predators and for social communication, and occur in greater numbers in shorter vegetation habitats (Laundré and Appel 1986, Michener and Schmutz 2002). Therefore, Unhealthy rangelands characterized by secondary succession plants and low vegetation are considered capable of supporting high densities of Richardson's ground squirrels (Fagerstone and Ramey 1996, Adams *et al.* 2010). The degree of grazing required to create or to maintain suitable habitat for Richardson's ground squirrels varies depending on vegetation type and moisture regime (Fagerstone and Ramey 1996). For example, moderate to heavy grazing may be required to open up habitat for Richardson's ground squirrels in productive mixedgrass or fescue prairie, whereas light grazing may be sufficient to maintain suitable habitat for ground squirrels in dry mixedgrass prairie with lower productivity and vigour and during drought conditions.

Other ground squirrel species have shown similar results as Richardson's ground squirrels to livestock grazing. In their study of California ground squirrel (*Spermophilus beecheyii*) response to livestock grazing, Howard *et al.* (1959) documented higher numbers of ground squirrels in fields grazed by livestock compared to ungrazed fields. Fehmi *et al.* (2005) found that ground squirrel occurrence and density did not differ between ungrazed and low to moderately grazed grassland and oak savannah habitats in California.

At high densities, Richardson's ground squirrels may potentially compete with cattle for forage resources due to the high amount of graminoids they consume (Fagerstone and Ramey 1996). However, the damage potential of Richardson's ground squirrels is often over-estimated and requires further research (Michener and Schmutz 2002). Given their small size and the fact that they are only active for a restricted period of the year, Michener and Schmutz (2002) estimated that it takes 2,275 Richardson's ground squirrels to equal the forage intake of one adult cow. Furthermore, research indicates that only 4% to 7% competition occurs between a similar ground dwelling rodent, prairie dogs, and livestock (Miller *et al.* 1994, Michener and Schmutz 2002). Other studies have found no significant difference in the market weight of steers that have lived with or without prairie dogs (Miller *et al.* 1994).

Michener and Schmutz (2002) caution that managers should conduct an adequate cost-benefit analysis to assess how much time, energy and money should be invested to control Richardson's ground squirrels that will ensure an appropriate return on their investment. This requires a scientific or economic study to establish: (i) the degree of forage competition between cattle and Richardson's ground squirrels; (ii) the extent of pasture damage caused by ground squirrels; and (iii) the frequency with which livestock are injured as a result of ground squirrel burrowing activity (Michener and Schmutz 2002). The ecological benefits of Richardson's ground squirrels should also be considered in this analysis, including their role as a fundamental prey species and the benefits of their burrowing activities to other fauna. The benefits of Richardson's ground squirrel burrowing activities also includes reducing soil compaction, and promoting soil aeration, nutrient cycling and soil development (Fagerstone and Ramey 1996). The potential for Richardson's ground squirrels to improve forage quality for cattle at colony sites, and selective grazing by cattle at colony sites should also be investigated (Coppock *et al.* 1983). Koford (1958) suggested that there was a reciprocal ecological relationship between bison and prairie dogs, whereby each maintained ideal habitat for the other, comprised of shortgrass species interspersed with patches of forbs and bare ground. Coppock *et al.* (1983) confirmed that bison fed selectively on moderately grazed, grass-dominated areas near the perimeters of prairie dog colonies, despite the fact that maximum plant biomass occurred on uncolonized sites. Bison were attracted to graze near prairie dog colonies as these areas had more readily digestible vegetation with higher crude protein and less dead vegetation and greater accessibility of green tissues than vegetation from uncolonized areas (Coppock *et al.* 1983). Where Richardson's ground squirrel population control is considered necessary, control strategies other than conventional poisoning programs that do not detrimentally impact non-target species such as swift fox are recommended (Michener and Schmutz 2002).

One strategy suggested for prairie dog and ground squirrel management is to use deferred grazing or reduced stocking rates to encourage greater amounts of taller and denser vegetation (Fagerstone and Ramey 1996, Michener and Schmutz 2002). Greater vegetative cover can make an area less suitable or less attractive to ground squirrels and may therefore provide a long-term method of improving rangeland productivity and maintaining ground squirrel populations at acceptable levels (Michener and Schmutz 2002). Establishing appropriate stocking rates and controlling the distribution and extent of moderately to heavily grazed areas can therefore be used as tools to manage availability of suitable ground squirrel habitat. Promoting raptors or American badgers as natural biological control agents is another strategy (Michener and Schmutz 2002). Determining acceptable ground squirrel population densities should account for the necessary abundance of ground squirrels needed to support a specified predator composition and abundance.

6 LIVESTOCK GRAZING SYSTEMS AND BURROWING MAMMAL HABITAT MANAGEMENT

Table III-10 provides an overview of five grazing systems and their potential positive and negative implications for maintaining or enhancing habitat for swift fox, American badgers and Richardson's ground squirrels. A grazing system is a tool used to control the spatial distribution, timing, intensity, and frequency of livestock grazing (Holechek *et al.* 2003). Applied research is

needed to properly assess the effectiveness of various grazing systems for providing habitat for these species in the Milk River and South Saskatchewan River watersheds.

Table III-10 Grazing Systems and Burrowing Mammal Habitat Management

Grazing System	Discussion
Continuous (Season-Long) Grazing	
Advantages:	Under a continuous grazing system, cattle have use of an entire grazing unit at once. The primary means of controlling the effects of grazing in this type of system involve setting stocking rates; utilizing water, salt or mineral placement and/or herding to distribute cattle; and controlling entry and exit dates. Under light to moderate stocking rates cattle will not utilize the full extent of the rangeland and a certain percentage of vegetation carryover will be retained. Areas of intense grazing pressure will be concentrated near water or salt or in patches with more palatable grasses. Formerly grazed patches will receive repeated use as these patches have a lesser build-up of litter and higher cover of more palatable regrowth vegetation (Robertson *et al.*1991). Other areas of the rangeland will receive moderate, light or no use depending on the palatability of forage, terrain and distance to stock water. Hence, conservative stocking rates can encourage patchiness with areas with heterogeneous amounts of use having variable structural and plant species composition characteristics. Richardson's ground squirrels are most likely to occur in highest densities in areas that receive the most use and in formerly heavily grazed patches or disturbance areas (Michener 2002). Northern pocket gophers and voles, in contrast, are more likely to occur in areas with greater vegetation cover (Fagerstone and Ramey 1996). Vegetation heterogeneity as stimulated by variable grazing intensity is therefore important for promoting small mammal prey diversity for American badgers and swift fox (Steenbergh and Warren 1977 as cited in Fagerstone and Ramey 1996, Jones *et al.* 2003, Bylo *et al*. 2014). A diverse prey base is likely to provide a more stable food supply for predators, such as American badgers and swift fox, during periods of drought or severe winters and given the cyclic nature of many small mammal populations. Moderate grazing also stimulates plant species diversity which has been positively correlated with increasing the diversity and richness of arthropod species (Willoughby 1993, Jonas *et al.* 2002). Arthropods provide an alternate important prey base for swift fox during the summer and autumn periods. Variable vegetation heights and density across the rangeland is also important for promoting suitable denning areas for swift fox with sufficient cover (Pruss 1999, Uresk *et al*. 2003).

Disadvantages:	Continuous grazing systems with high stocking rates will promote large areas of short and relatively uniform vegetation characteristics, depending on the grassland community and local terrain. Under heavy stocking rates grazing becomes less selective, resulting in all plant groups (and not only favoured forage species) being grazed. Heavy stocking rates also result in a substantial reduction in carryover of litter material. These conditions will likely favour high densities of Richardson's ground squirrels and provide an ample prey base for American badgers in particular. However, over time, consistent heavy grazing has been found to reduce small mammal species diversity overall and to depress populations of most small mammals (Fagerstone and Ramey 1996). Therefore, heavy grazing over a large area and time may have a negative effect on availability of microtine prey for swift fox during the winter months (Brechtel *et al.* 1996). The effect of heavy grazing on small mammal populations depends on the type of grassland and the small mammal species affected (Fagerstone and Ramey 1996). For example, grazing has been demonstrated to have a more profound effect on small mammals in tallgrass prairie than in shortgrass prairie, where plant cover is already low (Fagerstone and Ramey 1996). The effect of continuous grazing systems with high stocking rates on swift fox and American badger denning habitat requires further research. Based on results presented by Pruss (1999) and Uresk *et al.* (2003) this type of grazing regime may reduce the height and density of vegetation preferred at swift fox dens.
Complementary Grazing	
Advantages:	Complementary grazing has the potential to benefit the stability and diversity of the small mammal prey base available to swift fox and American badgers by allowing for deferred use and improved health of native prairie. Deferred use of native prairie is permitted due to early season grazing on tame pasture, whose forages typically mature much earlier than native grasses. With deferred use, native prairie vegetation is allowed time to develop during the growing season and vegetation cover and density may be improved, which then has the potential to increase rodent species abundance and diversity (Steenbergh and Warren 1977 as cited in Fagerstone and Ramey 1996). Skinner *et al.* (1996) reported that deferred grazed native grasslands that were part of the Medicine Wheel Project in southeastern Alberta had the highest values for vegetative cover and were the most productive habitat type for small mammals. Tame pastures also offer suitable habitat for certain rodents such as deer mice. Skinner *et al.* (1996) reported that small mammal abundance (primarily deer mouse) was three times greater in tame pasture than in native prairie under various rotational grazing strategies. Deer mice commonly travel above the snow during the winter and thus provide a suitable winter prey source for swift foxes (Pattie and Fisher 1999)

Disadvantages:	Unless heavily grazed, the taller growth habitat of most tame pasture forage species, such as smooth brome (*Bromus inermis* ssp. *inermis*) or timothy (*Phleum pratense*), creates unsuitable habitat for swift fox and Richardson's ground squirrels. In general, Richardson's ground squirrels, swift fox and American badgers occur more commonly in native prairie habitats than in tame pasture or cropland (Cotterill 1997, Michener 2002, Scobie 2002, Downey 2003b, unpublished survey results, Downey *et al.* 2006). Therefore, the benefits of complementary grazing depend on whether marginal cropland can be converted to tame pasture or if tame pasture is available within the grazing operation. Converting native prairie to tame pasture to allow for complementary grazing is unlikely to benefit swift fox, American badger or Richardson's ground squirrel, and will result in the loss of valuable native prairie habitat.
Rotational Grazing – Rest Rotation and Deferred Rotation	
Advantages:	Rotational grazing systems are often considered advantageous to wildlife as they allow for controlled, periodic rest and recovery of the rangeland. MacCracken *et al.* (1985) reported a significant positive relationship between small mammal abundance and cover in sagebrush-grass rangeland in Montana, where all areas were managed under a rest-rotational grazing system. Under a rest-rotational system, those fields that receive a full season of rest will likely become less suitable for Richardson's ground squirrels, which will be encouraged to disperse into more recently grazed fields. By encouraging areas of recovered vegetation cover and density this type of grazing strategy can thereby offer a means to control Richardson's ground squirrel density and dispersion. As mentioned previously, overall rodent abundance and diversity has been found to increase with increased vegetation cover and density (Steenbergh and Warren 1977 as cited in Fagerstone and Ramey 1996). Therefore, the overall effect of rotational grazing could increase the diversity and possible stability of the swift fox and American badger small mammal prey base. Rotational grazing systems, by portioning cattle into smaller management units, also offers a means to distribute grazing use more evenly across the area, resulting in more uniform grazing effects on vegetation. An overall reduction of vegetation height over a larger area improves habitat suitability for Richardson's ground squirrels and may offer favourable foraging habitat for swift fox. To create a diverse rodent prey base for swift fox and American badgers, rotational grazing should occur in large enough pastures or with light enough intensity to allow cattle to forage selectively (Bylo *et al.* 2014). Rotational grazing should aim to create a mix of heavily, moderately and lightly grazed as well as ungrazed fields.

Disadvantages:	Vegetation heterogeneity, although possible under rotational grazing systems, can be reduced if cattle are grazed beyond the 50% utilization point (Kobriger 1980). More uniform use of a greater area diminishes refuge habitat for voles and northern pocket gophers, which are important swift fox and American badger prey items (Fagerstone and Ramey 1996). Uniform heavy use, however, could promote higher densities of Richardson's ground squirrels (Michener 2002). Implementation of rotational grazing systems that result in the creation of numerous small fields, even if the fields are dominated by native prairie, may diminish the quality of the habitat for swift fox due to their preference for large grassland patches (Marks 2005). A minimum field size of 125 ac (50 ha) is recommended (Marks 2005).
Rotational Grazing – High Intensity Grazing	
Advantages:	Intensive grazing for short periods of time opens up habitat for Richardson's ground squirrels, particularly in areas with taller grass cover or high build-up of standing dead litter. High densities of Richardson's ground squirrels offer an abundant food source for American badgers and swift fox.
Disadvantages:	High densities of Richardson's ground squirrels may result in greater forage competition between ground squirrels and cattle, potentially increased incidence of cattle injury due to greater numbers of burrows and an overall depletion in rangeland productivity. Higher densities of a perceived rodent pest often prompt managers to resort to poisoning. Swift fox, American badgers and numerous other predators can be negatively affected by consuming poisoned Richardson's ground squirrels or poisoned bait. Consistent heavy grazing over time has also been found to reduce the overall abundance and diversity of small mammals in an area (Fagerstone and Ramey 1996). This has negative implications, as discussed, for reducing American badger and swift fox prey availability and stability, particularly during the winter.
Riparian Pasture Grazing	
Advantages:	Swift fox, American badger and Richardson's ground squirrels are all associated with open, flat, dry, upland grasslands. Riparian area grazing systems, therefore, may not have a significant effect on these three species. However, as productive riparian areas often harbour rich populations of insects and small mammals, these areas may serve as important source habitats for swift fox or American badger prey to disperse from (Strand and Merritt 1999). Studies have shown that small mammals in riparian habitats can be significantly detrimentally impacted by heavy grazing due to a loss of vegetative cover (Fagerstone and Ramey 1996). Riparian area grazing systems can benefit vegetation retention or improved vegetation cover in these areas and thereby serve to increase small mammal population sources.

Disadvantages:	Riparian area grazing systems attempt to control the season and amount of time cattle spend in riparian areas. If this results in a greater impact to upland grasslands, it may diminish habitat opportunities for voles or northern pocket gophers in these areas.

7 BENEFICIAL MANAGEMENT PRACTICE RECOMMENDATIONS

Re-establishment of stable swift fox populations in the Milk River and South Saskatchewan River watersheds is partially dependent on protection and appropriate management of native prairie rangelands using cattle grazing as a habitat enhancement tool. Management practices designed to benefit swift fox need to consider the ecological importance of maintaining American badger and Richardson's ground squirrel populations and their associated burrows as well as maintaining sufficient densities of other small mammal prey.

The following general land-use and grazing recommendations offer a variety of means to protect or enhance swift fox, American badger and Richardson's ground squirrel habitat within the Milk River and South Saskatchewan River watersheds and throughout the Grassland Natural Region of Alberta. Further research is required to improve our understanding of these species and their habitat requirements as well as their response to grazing and various grazing management practices (see Section 8).

7.1 General Recommendations

Native Prairie Conservation and Restoration

- Protect remaining native prairie in Alberta from cultivation. Conservation of large tracts of native dry mixedgrass and mixedgrass prairie is considered especially critical for maintaining suitable habitat for swift fox (Brechtel *et al.* 1996). Further conversion of grassland to cropland could threaten the persistence of swift foxes in Alberta (ASFRT 2007).
- Remove marginal cropland from production, where possible, and seed with native mixedgrass or dry mixedgrass graminoids to enlarge suitable habitat for swift fox, American badgers and Richardson's ground squirrels. When seeding to native species, work with a knowledgeable Agrologist, Biologist or Reclamation Specialist to ensure selection of appropriate native plant species for the area being restored/reclaimed.
- Work with and support not-for-profit conservation groups, such as MULTISAR, Alberta Conservation Association (ACA) and Nature Conservancy of Canada (NCC), that work to conserve native grassland habitat on private land in southern Alberta.

Habitat Protection and Industrial Development Mitigation

- Abide by Alberta Environment and Parks (AEP) recommended set-back distances and timing restrictions for swift fox dens in the Grassland Natural Regions (GoA 2011). During the breeding and kit-rearing seasons (*i.e.*, from February 16th to July 31st), swift fox dens have a recommended set-back distance of 500 m for all levels of human

disturbance (*i.e.*, high, medium and low), including land surveying, seismic drilling, and road, pipeline and facility construction. During the rest of the year (*i.e.*, August 1st to February 15th), dens have a recommended set-back distance of 50 m, 100 m and 500 m for low, medium and high disturbance activities, respectively. Burrows greater than 11 cm wide with a clear tunnel that are situated within 800 m of known maternal dens should also be protected with the same set-backs (ASFRT 2007). Maternal dens should be considered potentially active for five years after the last known occupancy (ASFRT 2007).

- Conduct pre-development wildlife surveys to locate swift foxes and/or their dens in areas with suitable habitat or known swift fox populations. Ensure wildlife survey data is entered into the Fish and Wildlife Management Information System (FWMIS) database maintained by AEP.
- Use Protective Notations under the *Public Lands Act* to protect key swift fox habitat and dens.
- Enforce provisions under the *Wildlife Act* protecting swift fox dens from disturbance.
- Publicly acknowledge companies that follow the swift fox BMPs through a formal recognition program (ASFRT 2007).

Pest Control

- Discourage the use of poisoning programs to control Richardson's ground squirrels and other so-called pests (*e.g.*, striped skunks). These types of programs negatively affect both swift fox and American badgers due to direct poisoning or reduced prey availability (FWD 1990, Brechtel *et al.* 1996, Scobie 2002). Encourage alternative Richardson's ground squirrel population control strategies (*e.g.*, biological control agents such as American badgers and raptors). Investigate vegetation management techniques, such as retaining greater amounts of taller and denser vegetation cover to manage Richardson's ground squirrel populations at appropriate densities (Michener and Schmutz 2002). Downey *et al*. (2006) observed that Richardson's ground squirrels selected against native prairie with grass taller than 30 cm in height, whereas Proulx and MacKenzie (2009) suggested that 15 cm vegetation height was sufficient to deter colonization of agricultural fields (*i.e.*, pastures, hay fields and cropland) by Richardson's ground squirrels. The Pest Management Regulatory Agency recently adopted changes to the labelling requirements on sodium cyanide containers to inform landowners of the dangers of the poison to swift fox by: (1) providing a website showing the range of the swift fox and (2) requiring users to consult with Alberta Fish and Wildlife officers for approval to use the poison if they live within swift fox range (Pruss *et al.* 2008).
- Given the reliance of swift fox on American badger burrows, direct hunting, trapping or poisoning of the latter has the potential to limit the availability of suitable denning burrows for swift fox. Fewer available burrows reduces predator escape habitat and lowers the quality or suitability of an area for swift fox.
- Promote organic farming practices and discourage the use of pesticides or herbicides on tame pastures and cropland in the vicinity of swift fox dens (Cotterill 1997). Pesticides can reduce the availability of insect or rodent prey or result in bioaccumulation of toxins in prey species (FWD 1990, Cotterill 1997). Furthermore, contaminated prey may affect swift fox reproductive success and survival (Cotterill 1997).

- Consider undertaking temporary, selective and judicious use of coyote control programs (*e.g.*, re-locations) where that species' presence may be a limiting factor to swift fox survival (Pruss *et al.* 2008). Take appropriate measures (*e.g.*, landowner education) to ensure swift foxes are not mistakenly targeted during coyote predator control activities.

Road Mortality

- Limit road development and enforce speed restrictions in critical swift fox habitat. Vehicle collisions are a substantial cause of mortality among swift fox, particularly juveniles (Kamler *et al.* 2003, Ausband and Foresman 2007, Moehrenschlager *et al.* 2007a). Road mortality is also a substantial cause of death among American badgers (Messick and Hornocker 1981, Weir *et al.* 2004, Hoodicoff *et al.* 2009). Encourage sharing of access roads among oil and gas companies operating within swift fox range.
- Record swift fox mortalities and correlate these with traffic counts along roads and highways within the species' range (ASFRT 2007).
- Manage oil and gas vehicle traffic in critical swift fox range through the use of Area Operating Agreements (ASFRT 2007). Such agreements can specify timing and use restrictions to minimize conflicts with swift fox.
- Post signage warning of potential collisions with swift fox in high mortality areas (ASFRT 2007).

Public Awareness and Education

- Educate landowners and land managers about the ecological importance of American badgers, Richardson's ground squirrels, swift fox and other species at risk. American badger burrowing activity also has ecological benefits to numerous other fauna and has soil aeration and water penetration benefits that should be considered (Scobie 2002, Eldridge 2004). As with Richardson's ground squirrels, encouraging tolerance towards American badgers by ranchers requires a cost-benefit analysis approach. The benefits of American badgers as a free, biological control agent for maintaining reduced Richardson's ground squirrel populations should be weighed against the actual costs incurred due to injured livestock or damaged machinery from American badger burrows and the costs of badger control efforts.
- Offer incentives to landowners for maintaining swift fox populations on their land. Provide financial or other compensation to landowners that undertake swift fox habitat enhancements. Offer public acknowledgement and appreciation to landowners who undertake initiatives to benefit species at risk.
- Encourage landowners to report sightings of swift foxes and/or their dens to AEP and restrict their access to dens during the spring and summer breeding and pup-rearing seasons.
- Establish and maintain good relationships with landowners in southern Alberta whose land contains species at risk. Be willing to listen to the concerns of landowners.

Population Monitoring

- Continue to monitor swift fox numbers and trends in southern Alberta. Continue to conduct inter-jurisdictional population censuses every five years (ASFRT 2007).

Censuses and populations trends will help determine the need for additional reintroductions.

- Infrared cameras and lures may offer cost-effective alternatives to mark-recapture methods for monitoring swift fox populations over large areas (Stratman and Apker 2014).

7.2 Grazing Recommendations

Livestock grazing as a land-use is compatible with protection of large tracts of native prairie and can be used to mimic patterns of disturbance that would have existed under historical bison grazing regimes. Swift fox, American badgers and Richardson's ground squirrels all co-existed with bison on the Great Plains of North America prior to European settlement (Carbyn 1989, Fagerstone and Ramey 1996, Scobie 2002). Bison grazing patterns are thought to have stimulated a patchy environment with some areas of intensive grazing and other areas of light or no grazing (Carbyn 1989, Adams *et al.* 1994). This patchiness is thought to be beneficial for stimulating small mammal abundance and species diversity by creating a variety of micro-habitats for species with varying cover requirements (Fagerstone and Ramey 1996).

In order to refine grazing management recommendations for individual grazing operations, local ecological conditions and defined wildlife management and livestock production objectives need to be taken into account. Ecological conditions include factors such as vegetation type, terrain, grazing history and small mammal community composition. Management objectives include defining tolerable Richardson's ground squirrel densities that will not jeopardize rangeland productivity but will also sustain a desired predator population density. Appropriate grazing systems should be developed site-specifically for participating ranches as part of the MULTISAR program.

The following recommendations provide general goals for maintaining habitat for swift fox, American badgers and Richardson's ground squirrels while also maintaining rangeland productivity:

- Promote patch grazing and heterogeneous vegetation heights, with small areas of heavy and no use and larger areas of moderate and light use. This will ensure that patches of shorter vegetation are maintained to sustain Richardson's ground squirrel populations, while also retaining patches of taller vegetation to support microtine (vole) rodent populations and northern pocket gophers. Voles are an important winter prey species for swift fox (Carbyn 1995, Brechtel *et al.* 1996), while northern pocket gophers are the second most important American badger prey in Alberta (Scobie 2002).
- In a continuous grazing system, light to moderate grazing (25% to 50% biomass utilization) can be used to prevent uniform utilization of the range and to promote patchy grazing (Adams *et al.* 2005, 2013a,b).
- To promote vegetation patchiness on a field basis, use rotational grazing systems or complementary grazing as a means to control timing, rest frequency and intensity of use in each field. Promote heavier grazing use in tame pastures rather than native prairie, especially earlier in the season. Tame forage species usually have a taller growth habit that is limiting to swift fox and Richardson's ground squirrels.

- Use deferred or rest-rotational grazing to allow for rest in native prairie to improve vegetation cover in areas where Richardson's ground squirrel densities exceed tolerable thresholds (Michener and Schmutz 2002). Encouraging increased vegetation cover will reduce habitat suitability for Richardson's ground squirrels.
- In areas where swift fox occur, avoid intensive grazing systems with high density stocking rates over large areas. Manage for average vegetation heights of approximately 12 cm at preferred den and foraging sites (Uresk *et al.* 2003).
- Use upland stock water development and salt placement as tools to encourage better distribution of cattle over the landscape and to create controlled, predetermined areas with heavier use. Rotate stock water and salt to distribute areas of heavier use and allow for recovery.
- Adjust stocking rates during drought conditions. Depending on past use, vegetation type, productivity and vigour, light grazing may be sufficient to maintain suitably short vegetation for Richardson's ground squirrels and swift fox.

8 RESEARCH RECOMMENDATIONS

Few studies have researched the effects of grazing intensity and different grazing systems on swift foxes, American badgers and their key prey species, including Richardson's ground squirrels. To assist managers in refining BMPs, research is needed to determine which grazing systems are best suited to sustaining viable populations of these species while also maintaining sustained rangeland productivity. In addition to this type of research, information is also needed to better define the population trends, habitat requirements, predator/competitor interactions and foraging habits of each of these species. Key research needs are described below for each species.

8.1 Swift Fox

- Conduct an assessment of the effects of variation in yearly and seasonal precipitation and temperature on swift fox prey base and microhabitat selection (Uresk *et al.* 2003).
- Define the characteristics and seasonal variation of swift fox foraging habitat and microhabitat selection in Alberta and Saskatchewan.
- Conduct an evaluation of swift fox movement patterns, behaviour and food habits in southern Alberta during the winter.
- Evaluate the vegetation structure and plant species composition of preferred winter foraging areas.
- Evaluate the effect of snow depth on selection of winter foraging sites.
- Determine the effects of different grazing regimes on the abundance and availability of important winter prey species utilized by swift fox (Brechtel *et al.* 1996).
- Evaluate the effect of grazing intensity and duration on selection of preferred foraging sites in various seasons.
- Conduct additional research to quantify home range sizes and juvenile dispersal distances for swift fox in Alberta and Saskatchewan under various grazing regimes and stages of range health in native prairie.

- Evaluate swift fox use of tame pasture and cropland versus native grassland in varying stages of range health.
- Investigate the suitability of silver sagebrush shrublands as swift fox habitat in southern Alberta (Olson and Lindzey 2002a).
- Determine Richardson's ground squirrel and American badger densities in preferred swift fox habitats. Investigate whether high Richardson's ground squirrel densities results in higher local predator populations and consequently higher predator-induced mortality among swift fox.
- Conduct field validation of the HSI and RSF models developed for swift fox habitat use in southern Alberta (Downey 2004a, Moehrenschlager *et al.* 2007b, Gan *et al.* 2014). Refine the HSI and RSF models as new data becomes available.
- On a section level, research the relative habitat preferences of swift foxes, red foxes and coyotes to determine how land-use changes might potentially influence interactions among these three canid species (ASFRT 2007).
- Within the swift fox's range, determine the relative population densities of red foxes and coyotes that would lead to reduced mortality of swift foxes and reduce the likelihood of range exclusion due to coyotes (ASFRT 2007).
- Refine existing population viability models (Moehrenschlager *et al.* 2007b) once new census, habitat and other data becomes available.
- Focusing on diseases that affect sympatric species, assess the prevalence of canid diseases, their vectors and potential impact on swift fox survival and reproduction (ASFRT 2007).
- Determine the effects of oil and gas activity on swift fox habitat use and survival (Pruss *et al.* 2008).
- Examine the potential effects of climate change on the swift fox.
- Conduct genetic analyses of swift foxes to determine bottlenecks, variability, connectivity and dispersal distances in Canada and within isolated populations in the United States (Pruss *et al.* 2008).

8.2 American Badger

- Initiate an American badger population monitoring program in southern Alberta to better define population sizes and trends.
- Conduct further research to determine American badger home range sizes in Alberta and determine the influence of Richardson's ground squirrel density, range health and grazing intensity on variation in home range sizes.
- Determine relative American badger abundance in pastures managed under various grazing systems and in various stages of range health.
- Determine Richardson's ground squirrel population densities that are required to sustain a specified number of American badgers.
- Evaluate the vegetation structure characteristics at American badger breeding dens.
- Conduct a cost-benefit analysis to determine the actual costs of American badger burrows to ranchers. Factor into this analysis the benefits American badgers offer as Richardson's ground squirrel predators, as well as the importance of their burrows to endangered species.

- Evaluate the extent of road mortality on American badgers in southern Alberta.

8.3 Richardson's Ground Squirrel

- Monitor Richardson's ground squirrel populations in southern Alberta to determine long-term ground squirrel population trends and habitat preferences (Downey 2003a). Attempt to determine the primary causal factors that are responsible for large-scale fluctuations in Richardson's ground squirrel populations.
- Determine the level of grazing that is required to create favourable habitat for Richardson's ground squirrels in dry mixedgrass, mixedgrass and fescue prairie under various moisture regimes and range health classes.
- Determine the amount and type of forage consumed by Richardson's ground squirrels and the potential for competition between cattle and ground squirrels (Michener and Schmutz 2002). Calculate the potential for Richardson's ground squirrel herbivory to impact cattle weight gains.
- Determine the effect of Richardson's ground squirrel herbivory on plant species composition and overall rangeland productivity.
- Assess cattle foraging behaviour in areas where Richardson's ground squirrels occur to determine whether cattle selectively graze near ground squirrel colonies (Coppock *et al.* 1983).
- Assess whether Richardson's ground squirrel herbivory improves the palatability or quality of forage available to cattle.
- Conduct a broad assessment of the impact of Richardson's ground squirrel poisoning programs on non-target avian and mammal species.
- Carry-out a cost-benefit analysis to determine the amount of energy and money that should be invested in Richardson's ground squirrel control relative to recolonization rates, and actual costs due to forage competition or burrow-induced injuries to cattle (Michener and Schmutz 2002).
- Compare the effectiveness of natural predators as Richardson's ground squirrel population control agents versus poison application.
- Determine tolerable Richardson's ground squirrel densities from a joint wildlife management and agricultural land-use management perspective.

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C. PRONGHORN

1 INTRODUCTION

The purpose of this report is to summarize the ecology and habitat requirements of pronghorn (*Antilocapra americana*) in southern Alberta. The response of pronghorn to livestock grazing, grazing management systems and range improvements that may be used to enhance pronghorn habitat are also reviewed. Beneficial management practices, including grazing management recommendations, are then discussed to provide guidance on pronghorn habitat management. These recommendations apply specifically to the Milk River and South Saskatchewan River watersheds of southern Alberta as well as to the greater Grassland Natural Region as a whole (Natural Regions Committee 2006). Lastly, a summary of outstanding pronghorn research recommendations is provided.

1.1 Background

The pronghorn is endemic to North America and is the only living member of its family. Pronghorns have tan-coloured bodies with white rumps, chests and chins and blackish snouts and cheeks. Unlike true antelopes, pronghorns have branched horns that are shed on an annual basis. Pronghorns are built for running long distances, and have evolved to have large hearts, tracheas and lungs compared to their body size (Buechner 1950). The pronghorn occurs in the western part of the continent from southern Alberta and Saskatchewan south through the United States to northern Mexico.

The pronghorn is designated as a ‘Sensitive’ species under the Alberta General status listing (Government of Alberta [GoA] 2016). The species is not listed under the *Species at Risk Act* and has not been assessed nationally by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC). Pronghorn are listed as a species of ‘Least Concern’ by the International Union for Conservation of Nature and Natural Resources (IUCN) (Hoffmann *et al.* 2008).

Once thought to number more than 35 million, the species almost went extinct in the early 1900s, but has since rebounded to the point where there were an estimated 700,000 individuals alive in the late 2000s (Hoffmann *et al.* 2008). In 2014, the North American pronghorn population was estimated at 804,180 animals (Weaver and Gray 2014).

Pronghorn in Alberta have a restricted distribution and are highly susceptible to climatic fluctuations and loss of native prairie habitat (GoA 2016). Despite the species’ vulnerability, the pronghorn population in the province may be on the upswing, although this conclusion is based on limited data (Jones pers. comm.). The Alberta pronghorn population was estimated to be 12,003 individuals in 2011 and 13,283 individuals in 2013 (Walker 2012, Weaver and Gray 2015). Pronghorn buck to doe ratios reported for Alberta in 2011 and 2013 were 38:100 and 42:100, respectively (Walker 2012, Weaver and Gray 2015). Doe to fawn ratios were 29:100 and 30:100 for 2011 and 2013, respectively (Walker 2012, Weaver and Gray 2015). Harvest data indicates that 181 pronghorn were killed in 2011 and 617 individuals were harvested in 2013 (including both sexes) (Walker 2012, Weaver and Gray 2015).

1.2 Ecology

1.2.1 Breeding

Pronghorn does typically breed when they 16 to 17 months of age (Yoakum *et al.* 2014). In northern latitudes, the majority of breeding takes place from mid-September to early October (O'Gara 1968). Pronghorn does typically have two fawns, although one or three fawns are not uncommon (Buechner 1950). Fawns in Texas are born between the first week of April and late May or early June, which is three to four weeks earlier than in Oregon (Buechner 1950). Pronghorns appear to exhibit birth synchronicity as a strategy to minimize predation of fawns (Gregg *et al.* 2001). When first born, fawns are greyish-brown and blend in well with their environment. Fawns spend the majority of their first few days hiding in suitable cover (Buechner 1950). When fawns are four to six weeks old, they and their mothers gather in small herds. Weaning occurs at approximately four months of age. Pronghorn fawns experience naturally high mortality rates (approximately 50% to 80%), with mortality rates influenced by predation, population size and climate (*i.e.*, precipitation) (Yoakum *et al.* 2004, Jones and Yoakum 2010).

1.2.2 Diet

More than 200 pronghorn diet studies have been conducted, with detailed reviews provided by Yoakum 2004a and Yoakum *et al.* 2014). Pronghorn diets vary with geographic location and the availability of plants in their environment as well as the season. Pronghorn diets consist primarily of forbs and shrubs (Buechner 1950, Beale and Smith 1970, Mitchell and Smoliak 1971, Sexson *et al.* 1981, McInnis and Vavra 1987). Grasses are sometimes consumed, particularly short, fine, green bunch grasses less than 8 cm tall (Yoakum *et al.* 2014). Based on his review of the literature, Yoakum (2004a) found that pronghorn diets in the Grassland Biome consisted of 62% forbs, 19% grasses and 17% shrubs.

In the Brooks area of southeastern Alberta, pronghorn diets consisted of 51% forbs, 35% shrubs and 13% graminoids (Mitchell and Smoliak 1971). Pronghorn in the Brooks study area preferred forbs and graminoids in spring (*i.e.*, April to May) and forbs and shrubs in summer, fall and winter (*i.e.*, June to March). Pronghorn in Alberta consume a variety of species, including 52 different forb species, 12 different shrubs species and an unknown number of grasses and sedges. The five most important food items, which comprised 73% of the year-round diet of the Brooks population, included: silver sagebrush (*Artemisia cana*), pasture sagewort (*A. frigida*), graminoids, cushion cactus (*Coryphantha vivipara*) and buckbrush (*Symphoricarpos occidentalis*). In a second pronghorn population in the Pakowki Lake area, the same five forb and shrub species comprised 90% of their late fall and early winter diets (Mitchell and Smoliak 1971). This second herd also consumed more graminoids in fall and winter than the Brooks population. For both herds, cultivated grain was a minor component of their diets (Mitchell and Smoliak 1971). Silver sagebrush appears to be especially important forage for pronghorn in winter, especially during harsh years (Martinka 1967).

In Kansas, one study found that forbs constituted more than 90% of pronghorn diet during certain months of the year (*e.g.*, June and July) despite only comprising 5% of the grassland community species composition (Sexson *et al.* 1981). During May through September, pronghorn diets in this study contained an average of 95% non-grass species. Alfalfa (*Medicago*

sativa) was used by pronghorn to supplement their diet in early spring before native forbs had begun growth. Winter wheat (*Triticum aestivum*) comprised a substantial part of pronghorn diets in late autumn, winter and early spring (Sexson *et al.* 1981). In Oregon, pronghorn preferred woody sagebrush species (*Artemisia arbuscular* and *A. tridentata*) in fall and winter and a mixture of various forb species (*e.g.*, Hooker's balsamroot [*Balsamorhiza hookeri*], longleaf phlox [*Phlox longifolia*] and wallflower phoenicaulis [*Phoenicaulis cheiranthoides*] in spring and summer (McInnis and Vavra 1987). In Texas, pronghorn consumed forbs more than shrubs or graminoids at all times of the year (Buechner 1950). Forb consumption was greatest in spring and lowest in autumn (based on average foraging time). Shrubs constituted a higher proportion of the diet in the fall. In all seasons, graminoids constituted a low percentage of the diet (range: 1% in winter to 7% in fall).

1.2.3 Predators

Mammals are the primary predators of pronghorn throughout their range. In Alberta, coyotes (*Canis latrans*) account for the vast majority of fawn deaths, with a lesser number of kills by bobcats (*Lynx rufus*) (Barrett 1984). Predation in this study was highest among fawns aged 4 to 15 days of age, and fawn mortality averaged 50% annually. In Oregon, 84% of monitored fawns died during the study period, with coyotes accounting for 86% of deaths (Gregg *et al*. 2001). In Yellowstone National Park in the United States, coyotes accounted for nearly 80% of pronghorn fawn mortality, with 66% of total mortality occurring within two weeks of birth (Barnowe-Meyer 2009, Barnowe-Meyer *et al.* 2009). In addition to coyotes and bobcats, golden eagles (*Aquila chrysaetos*) also prey on pronghorn (Yoakum *et al.* 2004, Jones *et al.* 2015). Yoakum *et al.* (2004) state that predation is highest during the first 30 days following parturition.

1.3 Habitat Requirements

1.3.1 General

Pronghorn prefer native grassland, savannah and shrub-steppe habitats on flat to gently rolling terrain (Yoakum 2004a). Pronghorn are generally considered native prairie species due to their affinity for this habitat type (Yoakum 2004a). In Alberta, highest pronghorn densities are generally found in large expanses of open native prairie on flat to rolling terrain (Jones *et al.* 2015). In general, pronghorn prefer open areas with slopes of less than 10% and they tend to avoid slopes of greater than 20% (Yoakum 2004a). Summer and winter ranges are usually differentiated on the basis of snow accumulation, the availability of seasonal forage and sources of drinking water (Yoakum *et al*. 2014). In Texas, pronghorn prefer rolling hills, which provide protection from winter storms and winds as well as suitable spring fawning grounds (Buechner 1950). Pronghorn there prefer large expanses of treeless terrain, although they will use trees for shade if they are readily available.

Pronghorn prefer low vegetation structure, ranging from 25 cm to 46 cm in height, and typically avoid or infrequently use vegetation greater than 64 cm in height (Yoakum *et al.* 2014). Tall vegetation reduces mobility and visibility, and therefore affects pronghorn survival (Goldsmith 1990). In Alberta, pronghorn are most common in the Dry Mixedgrass and Mixedgrass Natural Subregions, which are generally characterized by short and mid-height prairie grasses, forbs and shrubs (NRC 2006, Adams *et al.* 2013a). In their study of pronghorn populations in the Pakowki

Lake and Books areas, Mitchell and Smoliak (1971) described the vegetation in their study area, which was characteristic of native grassland plant communities in the Dry Mixedgrass Subregion (Adams *et al.* 2013a). In the Pakowki Lake area, preferred pronghorn habitat was described as a needle-and-thread – western wheat grass – blue grama (*Hesperostipa comata – Pascopyrum smithii – Bouteloua gracilis*) community type. Similarly, the Brooks pronghorn population was observed primarily on a needle-and-thread –blue grama (*Hesperostipa comata –Bouteloua gracilis*) community type.

Recent work by Jones *et al*. (2015) suggested that not all pronghorn select for native prairie habitat. Their study of pronghorn in Alberta and Saskatchewan showed that pronghorns generally fit into one of three groups: (1) those animals that preferred native prairie; (2) those that preferred cultivated areas (annual and perennial cropland); and (3) those that preferred a mix of native prairie and cultivated areas. Barrett (1982) found that pronghorn in Alberta used cultivated areas nearly 15% of the time throughout much of the year, except in October and November (late fall) when it increases to 25% of the time.

Jones *et al.* (2015) developed Resource Selection Function (RSF) models for pronghorn habitat use during fawning and winter in southern Alberta and Saskatchewan. Models for both fawning and winter habitat use by the three groups of pronghorns mentioned above (*i.e.*, native prairie, cultivated and mixed preference groups) all had the same five variables: land cover, land form, distance to express highways, distance to arterial and distance to collector roads. The three groups of pronghorn each had slightly different responses for the five variables. For example, in terms of land cover, the native prairie and mixed groups were more likely to select for native grassland habitat than the cultivated groups, which preferred cropland (both annual and perennial). In terms of land form, the native prairie groups selected for upper slopes, gentle inclines and constrained valleys; the mixed groups selected for upper slopes, small hills, shallow drainages, large hill tops and constrained valleys in relation to plains; and the cultivated group selected for small hills and shallow drainages in relation to plains.

Roads and fences are important determinants of pronghorn movements on the landscape (Sheldon 2005, Gavin and Komers 2006, Kolar 2009, Gates *et al.* 2012, Jones *et al.* 2015). Roads and highways act as filters or barriers to pronghorn movements across the landscape (Yoakum 2004b). Pronghorn show a non-linear response to roads, preferring generally to avoid them if possible, but also remaining near them for periods of time when they act as barriers to movement (Jones *et al.* 2015). Traffic intensity also has a negative effect on pronghorns, with higher traffic levels resulting in lower amounts of time foraging in spring (Gavin and Komers 2006). Jones *et al.* (2015) found that pronghorn generally foraged near low-traffic collector roads during the fawning period (*i.e.*, mid-May to mid-June) and mostly avoided all types of roads (including express highways, arterial roads and collector roads) in winter.

Although adapted to the dry prairie environment, adequate water sources appear to be an important determinant of pronghorn use of the landscape, at least in parts of the United States (Sundstrum 1968, Ockenfels *et al.* 1994, Clemente *et al.* 1995). Clemente *et al.* (1995) found that in New Mexico pronghorns were generally found within 2.6 km of a water source, and that individuals ranged further from water in summer than at any other time of the year. Pronghorn

will drink from springs, creeks, rivers, lakes, reservoirs and even troughs and other stock water developments (Sundstrum 1968).

1.3.2 Fawning Sites

In Alberta, fawning typically occurs on gently rolling native prairie, although hilly areas and flat terrain are also used but to a lesser extent (Barret 1984). Although pronghorn fawns will sometimes bed in tame pastures and cropland, diverse native prairie habitat with silver sagebrush shrubs, small depressions and stands of grasses and forbs greater than 25 cm tall provides the best fawn bedding cover (Barrett 1981). In Alberta, sagebrush cover may not be as important for fawn bedding sites as in other geographic locations, such as Idaho and Montana (Pyrah 1974, Autenrieth 1976, Barrett 1981). However, this may have to do with fawn age, as fawns appear to bed closer to sagebrush shrubs when they are younger and progressively bed further away from the shrubs as they age (Barrett 1984). Fawns from the same doe appear to bed in separate locations for the first few days after first being born, an apparent adaptive response to promote survival of at least one fawn. However, by approximately 24 days of age the siblings are practically inseparable (Barrett 1984).

1.3.3 Summer Habitat

Numerous habitat models have been developed to try to assess and map pronghorn habitat during different seasons (Yoakum *et al.* 2014). Kolar (2009) developed RSF models for both summer and winter habitat use in North Dakota. Variables he studied included vegetation type (*e.g.*, native grassland, barren, cropland, perennial grasses [tame pasture], forest, riparian, *etc.*), topography (ruggedness) and distance to primary, secondary and tertiary roads. In terms of summer habitat use, pronghorns in his study area selected for native grassland, cropland and tame pasture vegetation types and avoided rugged terrain, forest and riparian vegetation, and primary and secondary roads.

Dalton (2009) developed RSF models for pronghorn habitat use in southeastern Oregon, including on an annual basis, in spring and in winter. Her best-fit model included four variables: rock cover, normalized difference vegetation index (NDVI) (a measure of vegetation greenness that is correlated with nutritional quality), elevation and land cover. Pronghorn in her study selected for areas with higher NDVI and avoided high elevation areas, areas with high rock cover and communities with short-statured sagebrush (*Artemisia* spp.). The top spring model contained a single variable – cover of tall-statured sagebrush – which pronghorn selected for at that time of year.

1.3.4 Winter Habitat

Although pronghorn generally inhabit similar summer and winter habitats, there appear to be some slight differences. As previously discussed, Kolar (2009) developed RSF models for both summer and winter pronghorn habitat in North Dakota. The best-fit model describing habitat use in winter suggested that pronghorns avoided rugged terrain, forests and secondary roads and selected for native grassland and barren vegetation types as well as cultivated fields (including tame pasture).

Habitat suitability index (HSI) models developed for pronghorn winter habitat in Wyoming initially suggested that five variables were important in determining ideal winter range: shrub cover, shrub height, shrub diversity, topography, and availability of winter wheat fields (Allen and Armbruster 1982). Subsequent statistical analysis suggested only shrub cover, topography and shrub height were significant among those five variables (Irwin and Cook 1985). Field verification of the model resulted in a final best-fit HSI equation that incorporated the above five variables in addition to cover of all herbaceous species combined (Cook and Irwin 1985).

Beckmann *et al.* (2012) developed RSFs for pronghorn winter habitat use in the sagebrush-steppe of western Wyoming. Their models showed that pronghorn selected for sagebrush habitat over riparian areas, cropland or other types of vegetation. Pronghorn in this study also selected for lower elevation sites with lower amounts of snow cover. In comparison to flat terrain, pronghorn appeared to favour northeast, southeast and southwest aspects (Beckmann *et al.* 2012). Dalton's (2009) best-fit RSF model for pronghorn habitat use in winter contained one variable: low slope. Pronghorn in Oregon at that time of year selected for gentler slopes.

Torbit *et al*. (1993) found that pronghorn in their Colorado study area grazed in cultivated fields (winter wheat) in winter (November to April) and then switched to native prairie in summer (May to October). Jones *et al*. (2015) observed some pronghorn remaining in cultivated areas on a year-round basis, although these were not the majority of animals. Snow depths exceeding 25 cm to 30 cm may cause pronghorn to move to new areas due to the difficulty in obtaining forage (Yoakum *et al.* 2014).

1.3.5 Migratory Habitat

In his study of migrating pronghorn in Alberta, Saskatchewan and Montana, Jakes (2015) observed that animals selected for grasslands, intermediate slopes and south-facing aspects compared to other land cover types and topographic positions. In addition, pronghorn highly avoided sites with high densities of wellsites and road densities compared to sites with lower densities of these anthropogenic features. There were also season variations in pronghorn habitat preferences, with animals selecting for large hydrologic systems in fall and high quality forage areas in spring. Migratory pronghorn in the Northern Great Plains migrate south in fall after plant senescence, and migrate north in the spring selecting grassland sites with higher forage productivity (Suitor 2011).

1.3.6 Area Requirements

Pronghorn home range sizes appear to vary widely depending on geographic location. Apparently, size depends on habitat quality and quantity and, in some areas, seasonal movement and/or migration corridors to avoid deep snow (Yoakum *et al*. 2014). In Alberta and Saskatchewan, Suitor (2011) calculated mean summer and winter range areas for pronghorn at 5,840 ha and 13,730 ha, respectively, for a total area of 19,570 ha. Ockenfels *et al.* (1994) reported that pronghorn home range sizes varied from 4,300 ha to 28,730 ha in Arizona, while Dalton (2009) reported home range sizes ranging from 4,353 ha to 25,383 ha in Oregon, with an average of 12,000 ha. Clemente *et al.* (1995) found that pronghorn yearlings have larger home ranges in New Mexico than adults (3,400 ha vs. 1,400 ha), with values ranging from 938 ha to

3,773 ha for all age ranges. Males and females in this study did not differ in their home range requirements, and home range size did not differ by season.

Pronghorn also undertake daily and seasonal movements within and between home ranges. One study found that pronghorn range through up to 4.8 km^2 of habitat on a daily basis (Hjersman and Yoakum 1958, cited in Pollak 2007). Pronghorn daily movements increase progressively with age (Barrett 1984). By 25 days of age, pronghorn fawns regularly travel more than 1 km per day (Barrett 1984). In his study of seasonal movements of pronghorn in the Northern Great Plains, Jakes (2015) observed that pronghorn migrating in spring had mean and median Euclidian distance movements of 77.1 km and 56.7 km, respectively. In fall, pronghorn had mean and median Euclidian distance movements of 64.6 km and 54.6 km, respectively.

1.4 Pronghorn Response to Livestock Grazing

Pronghorn and cattle tend to co-exist peacefully on rangelands in North America (Yoakum 1975, Autenrieth *et al.* 2006). As browsers and grazers, respectively, pronghorn and livestock generally have different preferred food requirements. McInnis and Vavra (1987) studied the dietary relationships between cattle, horses and pronghorn in Oregon and found that there was low dietary overlap between pronghorn and either horses or cattle. Dietary overlap ranged from 7% in summer to 26% in winter for pronghorn and horses, and from 8% in winter to 25% in spring for pronghorn and cattle. Horses and cattle showed a high preference for grasses in this study, whereas pronghorn preferred shrubs and forbs. Therefore, pronghorn and cattle/horses are typically not competitors for the same food sources given resource abundance. In New Mexico, Howard *et al.* (1990) recorded dietary overlaps of only 16% between cattle and pronghorn. In a review of nine studies of cattle and pronghorn diets, Yoakum and O'Gara (1990) found that dietary overlap never exceeded 30%. However, pronghorn may switch to perennial grasses during periods of drought, and therefore directly compete with cattle and horses for food resources (Vavra and Sneva 1978). Therefore, it can be concluded that pronghorn and cattle are not major competitors for food resources on Healthy rangeland under normal climatic conditions. Instead, competition is most likely to occur in Unhealthy rangeland, in areas with a paucity of forbs and shrubs and in monocultures (Autenrieth *et al*. 2006)

In the Chihuahuan Desert of New Mexico, Smith *et al.* (1996) observed higher numbers of pronghorn on good compared to excellent condition rangeland, an observation the authors attributed to the lower diversity in vegetation structure and composition of the latter. Excellent condition rangeland had similar forb cover to good condition rangeland, but shrub cover was significantly lower. Clemente (1993) found similar results, with pronghorn using good condition rangeland more than ranges in lower or higher ecological condition. Similarly, Kindschy *et al*. (1982) suggested that pronghorn thrive best on mid-seral rangeland created by fire and herbivory by wild and domestic ungulates. In the Grassland Natural Region of Alberta, forb cover and diversity tends to be highest in light to moderately grazed grassland plant communities (Adams *et al.* 2005, Adams *et al.* 2013a,b). Loeser *et al.* (2005) suggested that removing cattle from Arizona rangeland does not lead to immediate improvements in forb cover and diversity and pronghorn fawn hiding cover

In Texas, pronghorn appear to fair well on over-grazed rangeland, albeit with altered diets (Buechner 1950). There, over-grazing leads to an increase in forb cover at the expense of perennial grasses. Although pronghorn favour the forbs found in good condition rangeland, they are able to alter their diets to survive on the weedier, poorer quality forbs on over-grazed rangeland. However, pronghorn in this area tend to weigh less on over-grazed compared to good condition rangeland. Therefore, over-grazing is likely not compatible with long-term pronghorn production.

Unlike cattle, pronghorn and sheep are direct competitors for forage. Sheep and pronghorn have much higher dietary overlap than pronghorn and cattle (Buechner 1950, Howard *et al.* 1990, Yoakum and O'Gara 1990). Howard *et al.* (1990) found that pronghorn and sheep have a dietary overlap of nearly 40% in New Mexico. Pronghorn tend to avoid rangeland grazed by sheep due to fact that sheep will consume forbs as well as graminoids and therefore leave little forage available for pronghorn (Buechner 1950). Pronghorn found on sheep pastures in New Mexico had much lower viability than those animals found on pastures grazed by cattle (Howard *et al.* 1990). Howard *et al.* (1990) suggest that in pastures where both sheep and pronghorn graze, sheep stocking rates will have to be drastically reduced so as not to harm pronghorn.

1.5 Grazing Systems and Pronghorn Habitat Management

Perhaps because of the aforementioned lack of dietary overlap between cattle and pronghorn, little research appears to have been conducted on the effects of different livestock grazing systems on pronghorns and their habitat. Moreover, of the literature that has been published, there appears to be little consensus on the effects of cattle grazing on pronghorn. Much of what follows is based on this limited understanding and has been inferred from the known effects of the different grazing systems in comparison to the known habitat preferences of pronghorn. The main point appears to be that cattle stocking rates are more important than the grazing system in their effects on pronghorn. The potential positive and negative implications of five grazing systems (continuous, complementary, rotational, deferred and intensive grazing) on pronghorn habitat are discussed in Table III-11.

Table III-11 Grazing Systems and Pronghorn Habitat Management

Grazing System	Discussion
Continuous (Season-Long) Grazing	
Advantages:	Provided proper utilization rates are observed whereby light to moderate grazing intensity is achieved, theoretically continuously grazed pastures can be managed in a manner that provides sustainable forage for both pronghorn and cattle. To counter some of the potential negative impacts of continuous grazing, such as overuse of some areas and underutilization of others, strategies such as salt and water placement can be used to encourage cattle away from critical pronghorn habitat (such as critical winter range).

Disadvantages:	Because pronghorn appear to favour good condition (Healthy) rangeland with abundant forbs and shrubs (Clemente 1993, Smith *et al.* 1996), continuous grazing that results in high utilization in all or certain parts of a field will reduce the suitability of that pasture for pronghorn.
Complementary Grazing	
Advantages:	Complementary grazing allows for deferment of native prairie-dominated fields during the fawning season, which may benefit pronghorn (Ockenfels *et al.* 1992). If complementary grazing leads to improved range health of degraded native pastures, this will further benefit pronghorn due to their preference for good condition (Healthy) native prairie (Smith *et al.* 1996).
Disadvantages:	Although pronghorn will occasionally make use of tame pastures (Sexson *et al.* 1981), native prairie broken and converted to tame pasture will reduce quality habitat and not benefit pronghorn due to their general preference for native grassland (Yoakum 2004a).
Rotational Grazing – Rest Rotation	
Advantages:	Rotational grazing often leads to more uniform grazing in a pasture. If this causes livestock to graze in previously ungrazed areas, it could lead to stimulated forb production. Rest-rotation grazing systems that allow for whole seasons of rest are likely to lead to improved range health of previously degraded pastures, thereby improving pronghorn habitat. Rest-rotation grazing that allows for deferment of native prairie-dominated fields during the fawning season will likely benefit pronghorn (Ockenfels *et al.* 1992).
Disadvantages:	Rotational grazing systems may require higher input costs due to the need for more fencing, water developments and/or herding. And if fences are not permeable to pronghorn, then they may create new barriers to movement. These costs may be balanced by improved range health and the potential to increase stocking rates as well as potential for higher sustainable pronghorn carrying capacities. Monitoring range health and adjusting stocking rates are important to ensure light to moderate use. As cattle are forced to be less selective in their forage preferences, cattle may make higher use of forbs under rotational grazing systems. Proper stocking rates can minimize use of non-preferred forage. In rest-rotation grazing systems, stocking rates have to be adjusted to account for the ungrazed field(s); otherwise, over-grazing could result, rendering the habitat less suitable for pronghorn.
Rotational Grazing – Deferred Rotation	
Advantages:	Ockenfels *et al.* (1992) suggested that deferring livestock grazing in areas with pronghorn fawning would help with recruitment by retaining hiding cover for newborn fawns.

Disadvantages:	Ranchers will need to find alternative feed sources during the spring to early summer period.
Rotational Grazing – High Intensity Grazing	
Advantages:	Intensive grazing systems with moderate stocking rates, sufficient rest and frequent monitoring may be suitable for stimulating a diversity of forb and browse forage for pronghorn.
Disadvantages:	Intensive cattle grazing before or during fawning season may displace females, leading to lower fawn recruitment (Yoakum and O'Gara 2000).

1.6 Range Improvements and Pronghorn Habitat Enhancement

As previously discusses, fencing can be a serious barrier to pronghorn movements on the landscape. Therefore, it reasons that fencing adjustments can be used to improve pronghorn habitat. Fencing recommendations to benefit pronghorn are discussed below (Section 1.7.2).

Use of cattle distribution tool such as low-stress herding and strategic supplement placement can be used to increase use of under-utilized grassland and, thus, improve pronghorn habitat by increasing forb production (Pollak 2007).

Pronghorn will readily use stock water developments such as developed springs with troughs in parts of the Unites States (Sundstrum 1968). Therefore, development of additional water sources in areas where few exist may lead to improved habitat conditions for pronghorn, although it is not clear if such improvements will benefit pronghorn in warmer northern climates such as Alberta.

1.7 Pronghorn Beneficial Management Practice Recommendations

The following recommendations provide a variety of means by which to enhance pronghorn habitat in the Milk River and South Saskatchewan River watersheds and throughout the Grassland Natural Region as a whole. Provided below are both general recommendations and recommendations related to grazing management.

Predator control to benefit pronghorns is generally not recommended. Control of predators such as coyotes can benefit pronghorn under certain circumstances (*e.g.*, where predators are a substantial limiting factor and the pronghorn population is below its carrying capacity), but the results are often short-term and the continued implementation expensive compared to habitat enhancements (Yoakum *et al.* 2004, Wakeling *et al.* 2015). Moreover, while coyote control programs may increase fawn survival, they may not increase herd size (Yoakum *et al.* 2004). Conservation of high quality habitat and use of habitat enhancements are likely more effective pronghorn management strategies (Yoakum *et al.* 2004).

1.7.1 General Recommendations

- Conserve large expanses of native prairie (Gates *et al.* 2012). Although cultivated areas may be utilized during all or certain times of the year (Jones *et al.* 2015), pronghorn generally prefer native prairie to other habitat types. Conservation of native grassland will also benefit many other species at risk in Alberta.
- Pronghorn-friendly fences can be effective, long-lasting tools for conserving pronghorns (Wakeling *et al.* 2015) and should be implemented where practical and feasible. For example, fences along the boundary of Canadian Forces Base (C.F.B.) Suffield were retrofitted with wildlife-friendly fences to better allow pronghorn movement (Gates *et al.* 2012). Similar action could be implemented elsewhere in the province where feasible.
- If necessary, employ selective use of road closures, land use restrictions and area closures to protect pronghorn populations during critical periods, including fawning and/or winter.
- Limit industrial and residential development in prime pronghorn habitat, including migration corridors (Seidler *et al.* 2015). Industrial development tends to reduce the quality of habitat available for pronghorn, with the species shifting its behavior to avoid such developments (Beckmann *et al*. 2012). In Wyoming, pronghorn did not travel within 100 m of natural gas well pads. Pronghorn probability of use dropped by 50% in parcels of land smaller than 300 ha created by energy development (Beckmann *et al.* 2011). Identification and protection of migration corridors and bottlenecks are crucial to conserving pronghorn populations (Sawyer *et al.* 2005). Roads associated with development may present barriers to pronghorn movements (Yoakum 2004b, Gates *et al.* 2012). Specific recommendations related to new oil and gas development include (Beckmann *et al.* 2011, Riley *et al.* 2012, Yoakum *et al.* 2014):
    - Minimize the drilling footprint by:
        - (1) Implementing phase-based development;
        - (2) Using directional (horizontal) drilling; and
        - (3) Using smart spatial configuration.
    - Implement winter closures to protect pronghorn winter habitat.
    - Ensure reclamation keeps pace with new developments. Reclamation of native grassland should follow the provincial guidelines for native prairie reclamation (Alberta Environment and Sustainable Resource Development 2013).
    - Implement remote monitoring of equipment and minimize the number of trips to site for other work to reduced disturbance caused by workers travelling to site.
    - Install gates along lease access roads to minimize public use.
    - Implement off-site habitat enhancements if habitat function around the site is impossible to maintain.
    - Implement special protection measures for pronghorn migration corridors.
    - Energy companies should be encouraged to share roads instead of building new ones.
    - Where fencing is necessary for oil and gas leases and rights-of way, ensure that the fences follow recommended guidelines for pronghorns (see discussion below).
- Wyoming Fish and Game Department (2010) has developed a comprehensive document detailing the mitigation measures required for energy development in pronghorn winter range based on the Category of Impact (moderate, high or extreme) or the development density (acres of disturbance per square mile). For example, at the extreme end of

development (*i.e.*, 16 well pads/mi^2 [6 well pads/km^2], or >80 acres of disturbance/mi^2 [12.5 ha/km^2]), proponents must implement seasonal use restrictions, habitat enhancement, compensatory off-site mitigation within the landscape planning unit and additional management prescriptions. Similar management prescriptions could be adopted by Alberta.

- Where feasible, construct over-passes or under-passes to allow movement across large highways, such as the Trans-Canada Highway, that may pose barriers to pronghorn migration. Roads and highways can be a serious barrier to pronghorn movement (Gavin and Komers 2006, Suitor 2011, Jakes 2015), and over-passes and/or under-passes may allow uninterrupted migration movement and gene flow between herds (McQuivey 2015). Over-passes may be preferable to under-passes but are likely more costly (Yoakum *et al.* 2014). Sawyer and Rudd (2005) provide the following recommendations for pronghorn underpass construction:
    - Large, open-span bridges along two-lane and divided multi-lane highways may serve as suitable crossing structures provided such structures allow for grade-level movements and unobstructed views of the horizon. The authors recommend a height of 5.5 m and a width of 18.3 m if the bridge is being designed specifically for pronghorn use. Column supports are also recommended over solid slab supports.
    - Existing open-span bridges can be modified to enhance their utility as pronghorn crossing structures by: (1) limiting human-related disturbance; (2) modifying or removing restrictive fencing; and (3) for those bridges that cross rivers, creeks or wetlands, expanding the length of the bridge to include upland habitat. Expanding the length of existing bridges during planned reconstruction or rehabilitation may be substantially less expensive than building new structures.
    - Inventory existing bridge structures that have potential to serve as crossing structures with minor modifications. Include in the inventory those bridges that cross waterbodies and riparian areas that could potentially be expanded. If possible, work with the Alberta Ministry of Transportation to make these modifications.
    - Investigate the effectiveness of concrete box and arch culvert underpasses in serving as pronghorn crossing structures.
- Work with the transportation industry and government transportation departments to increase of awareness of pronghorn needs and movements on the landscape (Gates *et al.* 2012).
- Work with the City of Medicine Hat to ensure future developments on the edge of the City remain permeable to pronghorn migration (Gates *et al.* 2012).

1.7.2 Fencing Recommendations

Fences are ubiquitous throughout the range of the pronghorn in southern Alberta. They are used to demarcate private land, Crown leases, oil and gas developments and highway rights-of-way. Pronghorn, if provided the choice, will migrate through areas with fewer or more permeable fences (Sheldon 2005, Gates *et al.* 2012). Unlike other ungulates such as deer (*Odocoileus* spp.), which will jump over fences, pronghorn prefer to crawl under or through fences, despite being

physically capable of jumping high enough to cross over top (O'Gara 2004, Harrington and Conover 2006).

Fences can have a number of potential negative effects on pronghorn, including:

- Direct mortality through entanglement (Yoakum 2004b). Fence material such as page wire may cause entanglement and ultimately death as the pronghorn try but fail to escape.
- Indirect mortality (Jones 2014); The barbs on the lower strands of barbed wire may cause scarring and hair loss along the backs, necks and hind quarters of pronghorn, possibly reducing fitness and potentially resulting in death (Jones 2014).
- Movement barriers (Suitor 2011, Gates *et al.* 2012). If the bottom wire is too low, it can prevent pronghorns from moving under the fence.

Because of the importance of fencing and its potential negative effects on pronghorn movement and survival, beneficial management practices specific to fencing are needed. A number of recommendations are provided below which should benefit pronghorn movement and survival in southern Alberta. While it may not be possible to implement all of these management actions on a particular ranch or lease, use of one or more of these techniques and tools will likely lead to some benefits for pronghorn.

- Limit new fence construction if possible and remove unneeded fences. Work with private landowners, Crown leaseholders, energy companies, other industrial players and government agencies to ensure fences do not present a barrier to pronghorn movement or a danger to individuals' survival. New fence construction in prime pronghorn habitat should ensure the movement needs of the species have been properly evaluated.
- Remove page (woven) wire fences and replace them with 3- or 4-strand horizontal wire fences (Paige 2008).
- A number of alterations can be made to make existing fences more permeable to pronghorn while still providing for livestock containment (Autenrieth *et al.* 2006, Paige 2008, Paige 2015, Sprague *et al.* 2012). First, the top and bottom strands of barbed wire should be replaced with smooth wire. In addition, the bottom wire should be moved up to a height of 40 cm to 45 cm to allow pronghorn to pass underneath. A minimum of 30 cm of space should be present between the top two wires. The top wire should be no more than 100 cm to 105 cm high.
- No more than two stays should be allowed between each set of posts, or if more than two stays are used, only two should be attached to the bottom wire (Yoakum *et al.* 2014).
- Avoid fencing natural water sources (*e.g.*, springs) important to pronghorn (Yoakum *et al.* 2014).
- If possible, leave gates to fields open when livestock are not present (Paige 2008, Gates *et al*. 2012, Paige 2015).
- Install let-down fences or adjustable/staple-lock fences at strategic pronghorn crossing locations (Yoakum *et al.* 2014). Let-down fences are designed to allow entire sections of fence to be detached from the rest of the fenceline and laid on the ground to facilitate pronghorn movement. Adjustable or staple-lock fences have adjustable bottom wires that can be raised or lowered depending on the situation. For such let-down and adjustable

fences to work, a person needs to be available at immediate notice to let adjust the fences when pronghorn are in the area (Yoakum *et al.* 2014).

- Where possible, move existing fences greater than 200 m from the edge of the highway (Sprague *et al.* 2012).
- If possible, work with landowners to remove fences along arterial and collector road rights-of way adjacent to ungrazed cropland or hay land. Fences along busy highways may present complete barriers to pronghorn movements (Seidler *et al*. 2012). Pronghorn may travel long distances parallel to fenced roads then cross when there is no road present (Sheldon 2005). Pronghorns may cross roads more often if fences are absent, and more crossings will ensure connectivity in pronghorn populations (Sprague *et al.* 2012). However, fences may still be needed along major highway to prevent wildlife-vehicle collisions (Clevenger *et al.* 2001). Therefore, serious consideration should be paid to all possible outcomes before a fence is removed. In such cases, highway under-passes or over-passes may be a better option (see discussion above).

1.7.3 Grazing Recommendations

The following list summarizes key livestock grazing management principles where the goal is to enhance pronghorn habitat:

- Grazing at a light to moderate intensity will promote growth of forbs and not negatively impact shrubs, which are the preferred foods of pronghorn. No grazing or over-grazing may create vegetation conditions unfavourable to pronghorn.
- While there appears to be little competition for food resources between pronghorn and horses or cattle, pronghorn and sheep are direct competitors. Rangeland grazed by sheep and pronghorn will need to have their sheep carrying capacities reduced in order to allow for sustained pronghorn production.

1.8 Research Recommendations

There are a few outstanding research questions primarily related to management and conservation, which are outlined below. In particular, little research has been conducted on the effects of livestock grazing on pronghorn habitat and the interactions between the two species. Answers to these questions will be of benefit to wildlife managers trying to conserve pronghorn populations in southern Alberta and elsewhere.

General research recommendations include the following:

- Determine the efficacy of area and road closures to humans in benefiting pronghorn during critical periods such as fawning (Yoakum *et al.* 2014).
- Evaluate the success of highway over-passes versus under-passes for allowing pronghorn movement across large, multi-lane highways.
- Using a modelling approach similar to Poor *et al.* (2014), determine the locations of high fence density areas within critical pronghorn habitat. Alternatively, look for other

approaches to spatially map fence lines to assess where fences are impeding pronghorn movement (Seward *et al.* 2012).

- Determine pronghorn migration pinch point locations to provide recommendations for underpass/overpass placement, especially across Highway 1 east of Medicine Hat (Jones pers. comm.).
- Scientifically assess the effectiveness of wildlife friendly fencing guidelines at creating permeable fences for pronghorn.

Research concerning the interaction between livestock and pronghorn is needed in the following areas:

- The relative use of native prairie habitat by pronghorn in Alberta under varying intensities or frequencies of cattle grazing.
- The effects of cattle grazing on the quality and sustainability of critical pronghorn winter range habitat.
- A comparative evaluation of various grazing systems (*e.g.*, complementary, rotational, deferred grazing) to improve pronghorn habitat and provide for sustainable grazing opportunities for ranchers.

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D. DEER GROUP (MULE DEER AND WHITE-TAILED DEER)

The purpose of this report is to summarize the ecology and habitat requirements of mule deer (*Odocoileus hemionus*) and white-tailed deer (*Odocoileus virginianus*) in southern Alberta in general, and more specifically within the Milk River and South Saskatchewan River watersheds. Inter-specific competition between deer species as well as the interaction and response of deer to livestock grazing is discussed. Grazing management systems and range improvements that may be used to enhance deer habitat are then reviewed and a summary of recommended beneficial management practices to manage deer habitat is provided. The recommendations apply to the Milk River and South Saskatchewan River watersheds as well as to the greater Grassland Natural Region of Alberta (Natural Regions Committee [NRC] 2006). Lastly, a summary of research recommendations is given.

1 INTRODUCTION

1.1 Background

Mule deer and white-tailed deer in the Milk River and South Saskatchewan River watersheds are economically important as big-game species for recreational hunters (Alberta Fish and Wildlife Division (FWD) 1989, Alberta Environmental Protection [AEP] 1995). Deer populations are regularly monitored in Alberta and hunting is controlled through season length, bag limits, sex restrictions and zoning (FWD 1989, AEP 1995). Managing deer populations effectively also requires an understanding of their habitat requirements and habitat enhancement strategies. Since deer co-exist with livestock throughout much of their range, several studies have attempted to examine how cattle grazing influences deer foraging behaviour as well as how it can be used as a tool to enhance deer habitat (Holechek *et al.* 1982, Kie *et al.* 1991, Loft *et al.* 1991, Telfer 1994). Livestock grazing has the potential to affect deer habitat by altering vegetation composition, productivity, nutritive quality and structure (Severson and Urness 1994).

Mule deer and white-tailed deer are both listed as ‘Secure’ in Alberta under the General Status listing (Government of Alberta [GoA] 2016). Neither species has been assessed nationally by the Committee on the Status of Endangered Wildlife in Canada and neither species is listed on the federal *Species at Risk Act* (COSEWIC 2016). Internationally, both the mule deer and the white-tailed deer are listed as species of ‘Least Concern’ by the International Union for Conservation of Nature and Natural Resources (IUCN) (Gallina and Lopez Arevalo 2008, Sanchez Rojas and Gallina Tessaro 2008).

Mule and white-tailed deer are most abundant in the prairies and parklands of Alberta, but are common throughout much of the province (FWD 1989, AEP 1995). Populations of both species declined to very low levels in the late 1800’s due to severe winters and high hunter harvest levels (FWD 1989, AEP 1995). In the 1900’s, the numbers recovered, peaking in the 1950’s then declining again in the 1960’s and early 1970’s (FWD 1989, AEP 1995). Both mule and white-tail deer populations within the Milk River and South Saskatchewan River watersheds experienced an increase in the early 2000s (FWD 1989, AEP 1995, Erickson 2002, Taggart pers. comm.). A series of mild winters and conservative harvest regimes are cited as the primary

factors for recent population increases (FWD 1989, AEP 1995). More recent population estimates are not available.

Aerial winter surveys are conducted either annually or on a rotational basis to monitor mule and white-tailed deer populations in various Wildlife Management Units (WMUs) in southern Alberta (Dube pers. comm.). The Milk River Watershed occupies all or a portion of WMUs 102, 104, 106, 108 and 116 and 624, whereas the South Saskatchewan River Watershed covers all or a portion of WMUs 102, 106, 112, 116, 118, 119, 124, 128, 144, 150, 624 and 732 (FWD 1989, AEP 1995). Survey information is used to calculate pre-season (*i.e.*, fall) population estimates for each WMU annually. The combined 2015 pre-season population estimate for the WMUs in the Milk River and South Saskatchewan River watersheds was 19,130 mule deer and 16,256 white-tailed deer (M. Grue pers. comm.).

1.2 Ecology

1.2.1 Breeding

The breeding (rutting) season for deer in Alberta usually occurs between late October and early December (Van Tighem 2001). Deer do not form harems as elk (*Cervus elaphus*) do; however, a successful, typically older buck may impregnate several females during the mating season (Van Tighem 2001). Fawns are usually born in May or early June. The gestation period lasts approximately six and one-half to seven months (Pattie and Fisher 1999, Van Tighem 2001). Does on average give birth to two fawns or twins, but may produce from one to three young (Pattie and Fisher 1999). Yearling does and does on poor range may produce single fawns (Pattie and Fisher 1999). Age at first parturition may be as early as one year if on good range, but two years is more common (Van Tighem 2001). Females are seasonally polyestrous (Peek and Krausman 1996, Teer 1996). A doe can undertake up to four cycles in a breeding season as the estrous cycle is repeated every 28 days, although they usually conceive in their first heat. Fawns are most often weaned at four to five months of age (Peek and Krausman 1996, Teer 1996). A doe and fawn will typically stay together until the next spring. Due to the overlapping ranges of mule deer and white-tailed deer in Alberta, hybrids have been observed (FWD 1989, AEP 1995). The survivability and fertility of hybrids is generally considered inferior to non-hybrids (AEP 1995, Van Tighem 2001). Hybrids in Alberta associate mainly with mule deer (FWD 1989). Deer may live up to 20 years; however most live less than 10 years (Peek and Krausman 1996, Teer 1996).

1.2.2 Diet

White-tailed and mule deer diets overlap considerably in Alberta (FWD 1989, AEP 1995). The winter and summer diets of deer tend to differ in composition (Gadd 1995, Pattie and Fisher 1999, Wood *et al.* 1999). In fall and throughout winter, deer browse on the leaves and twigs of various shrub and tree species (FWD 1989, AEP 1995, Pattie and Fisher 1999, Wood *et al.* 1999). Forbs and grasses make up a substantial part of their spring and summer diet (Wood *et al.* 1999). There is typically an energy surplus available to deer for synthesis and storage of fat during the summer, and an energy deficit during the winter (Wallmo *et al.* 1977). Winter forages contain less of the digestible components such as proteins, starches and sugars than summer forage (Wallmo *et al.* 1977).

Deer are primarily browsers, feeding on trees, shrubs and forbs (Findholt *et al.* 2004, Beck and Peek 2005). Common winter forb and browse species consumed by white-tailed and mule deer in Alberta include: asters; snowberry (*Symphoricarpos albus*, *S. occidentalis*); aspen (*Populus tremuloides*); rose (*Rosa* spp.); saskatoon (*Amelanchier alnifolia*); choke cherry (*Prunus virginiana*); willow (*Salix* spp.); silverberry (*Elaeagnus commutata*); and creeping juniper (*Juniperus horizontalis*) (Rhude and Hall 1977, FWD 1989, AEP 1995). Other important winter forb species for white-tailed deer include peavines (*Lathyrus* spp.) and horsetails (*Equisetum* spp.) (AEP 1995). Pasture sagewort (*Artemisia frigida*) has been identified as an important component of the mule deer winter diet in Alberta (FWD 1989). Both mule and white-tailed deer will also eat agricultural crops, such as alfalfa (*Medicago sativa*), winter wheat (*Triticum aestivum*), fall rye (*Secale cereale*), cultivated oats (*Avena sativa*) and cultivated barley (*Hordeum vulgare*) particularly during the fall and spring (FWD 1989, AEP 1995).

Although primarily browsers, under certain circumstances deer will also feed on graminoids (*i.e.*, grasses, sedges and similar plants). Bluebunch wheatgrass (*Pseudoroegneria spicatum*), Idaho fescue (*Festuca idahoensis*), intermediate oat grass (*Danthonia intermedia*), and Indian rice grass (*Achnatherum hymenoides*) are examples of grasses that are rated as good to excellent native forage for mule deer during the spring and summer (Peek and Krausman 1996). Of the common introduced forage grasses, timothy (*Phleum pratense*) provides good to excellent mule deer forage, while crested wheatgrass (*Agropyron pectiniforme*) is rated as fair (Peek and Krausman 1996). Stewart *et al.* (2003) reported that mule deer in Oregon consumed more sedges than any other plant group during summer, including conifers, shrubs, grasses and forbs, despite following what the authors described as a 'browsing strategy'. Torstenson *et al.* (2006) reported that mule deer diets in Wyoming in winter/spring contained 31% graminoids, 33% browse, 18% alfalfa and 18% forbs other than alfalfa, whereas in summer/fall their diets contained 16% graminoids, 27% browse, 24% alfalfa and 32% forbs other than alfalfa. Beck and Peek (20005) reported that mule deer diets in Nevada in summer were comprised of 64% to 72% forbs, 30% woody browse and 2% to 5% graminoids.

In comparison to other cervids (members of the Deer Family) such as sheep, goats and elk, deer may be more selective in their forage preferences and, therefore, have a smaller food base (Collins and Urness 1983). Deer have lower digestive efficiency and are less capable of digesting the lignified fiber of non-preferred grasses and sedges (*Carex* spp.). Consequently, deer may be less successful at competing with other cervids on poor condition (Unhealthy) range (Collins and Urness 1983). However, mule deer in Oregon had a more variable diet than either cattle or elk, indicating that deer there were opportunistic feeders (Stewart *et al.* 2003). More research is needed to full understand deer foraging preferences under different circumstances.

1.2.3 Predator Response

Primary deer predators include coyotes, cougars and humans (Van Tighem 2001). White-tailed deer and mule deer respond somewhat differently to predators (Lingle 2002). Mule deer are more adapted to open areas near steep terrain and tend to elude predators by moving to and up slopes (Van Tighem 2001, Lingle 2002). White-tailed deer appear to be more adapted to gentler terrain and rely on their greater speed and proximity to dense cover to evade predators (Van Tighem

2001, Lingle 2002). Unlike mule deer, white-tailed deer tend to move down and away from slopes toward cover when confronted by predators (Lingle 2002). The formation of herds is thought to be another adaptation to predation and is common to both species (Van Tighem 2001). Predators such as coyotes, for example, are more likely to attack and kill an individual deer when herds split and individuals become isolated (Lingle 2003). In southeastern Alberta, Lingle (2003) found that mule deer formed larger, more cohesive mixed-sex groups during the winter than white-tailed deer. Lingle (2003) suggests that herding may be more important as an anti-predator strategy for mule deer than for white-tailed deer as they are slower and less effective at fleeing from predators than white-tailed deer.

1.2.4 Movement Patterns and Herding

Mule and white-tailed deer both tend to move between winter and summer ranges (Kramer 1971a, Van Tighem 2001). In general, white-tailed deer in the prairies are considered less 'migratory' than mule deer and often occur in the same area year-round if conditions are suitable (Van Tighem 2001). Movements in winter are associated with access to better thermal cover and more accessible food resources (FWD 1989). Winter ranges for deer in the Milk River and South Saskatchewan River watersheds are more discrete than summer ranges. Seasonal deer movements vary in response to snow cover, temperature, access to food resources and human disturbance (FWD 1989, AEP 1995, Van Tighem 2001).

Mixed herds of does and bucks come together during the rut in the fall and during the winter (Van Tighem 2001). Snow depth and restricted availability of good quality forage mean that deer often congregate in groups in suitable winter range. When not constrained by winter snow, white-tailed deer prefer groups of no more than 10 individuals (Van Tighem 2001). During the spring and summer, mule and white-tailed deer bucks form small bachelor herds or become solitary, whereas does often associate with other does and their fawns following fawning (Van Tighem 2001, Lingle 2003).

1.3 Habitat Requirements

1.3.1 General

A combination of cover and feeding areas is critical for both mule and white-tailed deer (AEP 1995). A shrub-forest-meadow complex is ideal for deer in that it provides a combination of good forage selection and cover to provide protection from the elements as well as from predators (AEP 1995). Due to the limited availability of cover and water on the prairies, shrubby and wooded draws and forested river valleys provide important thermal cover, succulent vegetation, open water, travel corridors and diverse forage for deer (Nietfeld *et al.* 1985). Riparian and wetland zones are therefore heavily used by deer (Nietfeld *et al.* 1985).

1.3.2 Fawning Sites

In order to escape predation in early life, fawns rely on hiding cover and are considered hider-type neonates (Kie *et al.* 1991). Young deer are spotted and scentless and will hide under shrubs or trees or in tall grass for a month to nurse (Gadd 1995). A doe will cache their fawns separately, often under the lower boughs of spruce trees or in brush up to 100 m apart (Gadd 1995). Sites with dense shrub cover and proximity to forest cover and water, such as riparian

areas, are considered optimal fawning habitat (Wood *et al.* 1999). Woody draws, drainages and basins with slopes of less than 15% are commonly used for fawning cover by white-tailed deer in Alberta (Nietfeld *et al.* 1985). In the grasslands, mule deer use deciduous thickets for fawning sites (Nietfeld *et al.* 1985). Availability of water within 180 m is considered important during fawning (Nietfeld *et al.* 1985).

1.3.3 Summer Habitat

Deer are more versatile in their habitat use during the summer than during the winter as they are not limited by snow cover and can choose from a more varied, higher quality forage supply. Consequently, deer are typically more dispersed and occur in lower densities across the prairie landscape during the summer (Van Tighem 2001). Proximity to water may limit the extent of deer summer range during drought years (Nietfeld *et al.* 1985). The summer is a critical growth period for deer to restore energy reserves depleted over the winter months (Loft *et al.* 1991). The quality of summer habitat is important to ensure proper nourishment of deer in preparation for the winter (Scotter 1980). Early-season summer habitat is particularly important to maternal female deer that have a high nutritional demand and rely on high quality forage and cover to nourish and hide young fawns (Loft *et al* 1991).

In general, white-tailed deer tend to stay close to riparian areas year-round in prairie environments, as they are more adapted to dense cover (Van Tighem 2001). However, use of habitat types by white-tailed deer may vary slightly be gender; for example, in Texas Cooper *et al.* (2008) observed that female white-tailed deer made greater use of riparian and drainage areas with dense cover than bucks, which tended to be found in more open habitats. Mule deer typically use more open habitats, but do require some cover for daytime bedding (Nietfeld *et al.* 1985). Shelterbelts in agricultural fields, or sagebrush or shrubby draws in open prairie, provide adequate bedding cover in open areas (Taggart pers. comm.). Mule deer have been observed to use the same bedding areas over a summer, perhaps as a predator-defense behavior (Collins and Urness 1983). Torstenson *et al.* (2006) reported that mule deer in Wyoming made substantial use of alfalfa hay fields.

In their study of mule and white-tailed deer in Texas, Avey *et al.* (2003) found that five habitat variables best explained habitat use by the two deer species: slope; presence of shrubs; presence of forbs; presence of grass; and vegetation classification. Of these five variables, percent slope was the most important, with mule deer using mean percent slopes of 8.6% and white-tailed deer using mean percent slopes of 4.0%. Mean shrub cover used by mule deer and white-tailed deer for bed sites was 37.7% and 53.9%, respectively. Mule deer in this study were more commonly found on relatively steep slopes with moderate to sparse vegetation cover, whereas white-tailed deer most often occurred on flatter terrain with higher vegetation cover.

1.3.4 Winter Habitat

Snow depth is the critical factor that forces deer to migrate to suitable wintering ranges, and it determines the extent of suitable range available to deer (Scotter 1980). Generally, a snow depth of approximately 46 cm for mule deer, and snow depths of approximately 41 cm to 50 cm for white-tailed deer, prompt them to leave their favorite summer or early fall range and head towards winter ranges (Kramer 1971a). Deer in the Milk River Watershed reportedly move to

winter ranges when snow depths reach 15 cm to 20 cm (Taggart pers. comm.). Winter ranges used by deer in southern Alberta tend to be south- and west-facing, wind-swept grassy slopes that have low snow cover (Taggart pers. comm.). Areas with less snow improve access to forage and facilitate ease of movement.

In the Milk River Watershed, white-tailed deer are found in treed river valleys in the winter, whereas mule deer are more likely to use coulee breaks and slopes (Lingle pers. comm., Taggart pers. comm.). In addition to treed river valleys, coniferous shelter-belts and aspen clumps also provide winter thermal cover for both species (AEP 1995, Teer 1996). There may be a greater need for sufficient winter cover for white-tailed deer in southern Alberta as there is a higher winter mortality rate for this species (Taggart pers. comm.). The preference of white-tailed deer for habitats with a greater degree of woody cover, such as forests, shrublands and riparian forests, is well documented (Teer 1996). Forage to cover ratio of 60:40 is considered optimal for winter mule deer habitat (Wood *et al.* 1999). Mule deer in southern Wyoming and northern Colorado selected for steeper and shallower slopes (quadratic relationship) in winter as well as proximity to rocky outcrops and shrubland, and the species avoided roads and grassland habitat (Webb *et al.* 2013). Although the relationship was less influential, mule deer in this study also tended to be found closer to developments and agriculture during winter (Webb et al. 2013).

1.3.5 Area Requirements

Deer home range size depends on availability of food and cover and typically varies according to season (Nietfeld *et al.* 1985). Where adequate food and cover is available, home ranges tend to be small (Nietfeld *et al.* 1985).

White-tailed deer summer home ranges vary from 70 ha to 190 ha, while winter home ranges vary from 160 ha to 480 ha (Nietfeld *et al.* 1985). In areas of good habitat, white-tailed deer may remain in a 500 ha to 800 ha area (Nietfeld *et al.* 1985). For white-tailed deer in Alberta, the following optimum sizes of various cover types have been reported: thermal cover – 0.8 ha to 2.0 ha; hiding cover – 2.6 ha to 10.5 ha; and fawning cover – 0.4 ha to 2.0 ha (Nietfeld *et al.* 1985). Similar optimum sizes of thermal cover (0.8 ha to 2.0 ha) have been reported for mule deer (Nietfeld *et al.* 1985). In Sierra Nevada, California, female mule deer summer ranges averaged 88 ha in the absence of cattle grazing, and ranged from 103 ha to 124 ha with heavy grazing (Kie *et al.* 1991). In an agricultural environment in Minnesota, white-tailed deer had average home range sizes of 520 ha and 230 ha in winter and summer, respectively (Brinkman *et al.* 2005). Deer densities vary considerably across the landscape of the Milk River Watershed. High densities of mule deer occur in association with areas of shrubland and high topographic relief, such as coulees, sandstone complexes and badlands (Quinlan pers. comm.). The highest densities of white-tailed deer exist in the riparian habitats of major river valleys, often in association with cottonwood (*Populus* spp.) stands, and in agricultural areas (Quinlan pers. comm.).

1.4 Mule and White-Tailed Deer Interactions

As there is a high degree of dietary overlap between mule and white tailed-deer, the potential for competition between these species does exist (Rhude and Hall 1977). Topographical segregation

and differing anti-predator behaviour between mule and white-tailed deer may lessen opportunities for competition to occur (Swenson *et al.* 1983, Wood *et al.* 1989, Lingle pers. comm.). As mentioned previously, mule deer tend to prefer more rugged and open terrain, while white-tailed deer prefer gentler terrain with greater cover (Lingle pers. comm.). However, habitat selection differences between mule and white-tailed deer vary regionally, and little is known about resource partitioning between the two species where they co-occur (Avey *et al.* 2003). For example, Avey *et al*. (2003) found that vegetation structure played a more important role in habitat partitioning than topography. There is also potential for habitat overlap between these species in prairie areas where cover is limiting or during winters with high snow cover. In general, however, potential for competition is increased where forage is limiting, such as during drought conditions or in areas that are intensively grazed by cattle (Peek and Krausman 1996). Overall, Kramer (1971b) suggested that competition between white-tailed and mule deer is not significant in natural habitat. Similarly, Lingle (2002) reported similar feeding habits by mule and white-tailed deer with no evidence of competition in southeastern Alberta.

1.5 Mule and White-Tailed Deer Response to Livestock Grazing

Domestic livestock grazing has the potential to affect the quality of deer habitat by changing plant productivity or altering plant succession in a manner that either favours or reduces forage and cover (Peek and Krausman 1996). Deer may also be directly influenced by the presence of livestock on the landscape and associated human activity (Peek and Krausman 1996). The relationship between deer and livestock varies locally and depends on ecological conditions (*i.e.*, plant communities and productivity, topography and climate) (Peek and Krausman 1996). In general, there is low dietary overlap between deer and cattle (Stewart *et al.* 2003, Beck and Peek 2005, Torstenson *et al.* 2006); however, the degree of overlap varies with intensity of use of a range and is affected by drought conditions (Peek and Krausman 1996). Grasses typically dominate the summer diet of cattle, while deer consume a significantly greater proportion of browse and forbs (Loft *et al.* 1991, Peek and Krausman 1996, Torstenson *et al.* 2006). As cattle have proportionately larger rumens than deer, they are able to digest forage of lower quality (Peek and Krausman 1996). Potential for competition between deer and cattle is increased with higher cattle stocking rates and during years of below average precipitation (Kie *et al.* 1991, Peek and Krausman 1996). Under these conditions, the ability to partition resources is limited and competition is thereby intensified (Stewart *et al.* 2002). Potential for competition is also increased in arid environments due to limited cover and availability of water, and subsequently a shared preference for riparian areas (Holechek *et al.* 1982).

Various studies have demonstrated that livestock grazing can exert an influence on deer activity and distribution patterns, habitat use and population density (Kie *et al.* 1991, Loft *et al.* 1991, Telfer 1994, Peek and Krausman 1996). Kie *et al.* (1991) and Loft *et al.* (1991) studied the influence of grazing on female mule deer on summer range in Sierra Nevada, California. Cattle grazing reduced the availability of forbs preferred by mule deer, especially in late summer (Kie *et al.* 1991). To compensate, deer consumed a broader range of plants with lesser nutritive value, such as sedges (Kie *et al.* 1991). In addition, deer reportedly spent more time feeding and less time resting and increased their home ranges to include steeper slopes and less preferred habitat (Kie *et al.* 1991, Loft *et al.* 1991). Moderate to heavy livestock grazing reduced deer use of preferred meadow-riparian areas and excluded their use of aspen habitats due to reduced forage

and cover (Loft *et al.* 1991). The greatest potential for diet overlap and competition with cattle and mule deer occurred in late summer in mutually preferred meadow-riparian habitats when forage was at a minimum (Loft *et al.* 1991). Therefore, moderate to heavy livestock grazing can result in competition for forage and increase predation risks to deer by reducing hiding cover available to fawns and forcing deer to spend more time feeding (Kie *et al.* 1991, Loft *et al.* 1991). As deer are displaced from preferred habitats by heavy cattle grazing, this may also influence their nutrition, survival and productivity.

Telfer (1994) examined cattle and cervid (moose, elk and deer) interactions in a foothills watershed in southwestern Alberta. Cattle diets were most similar to elk diets, being comprised primarily of herbaceous material (grasses and forbs). Cattle diets consisted of 89% herbaceous material. Deer diets contained minor amounts (6.4%) of herbaceous matter and 43% browse (primarily from saskatoon shrubs and aspen). Deer selected the forest cover and browse of steep middle slopes, while cattle primarily used lower slopes (less than 20%) with high herbaceous cover. The tendency for cattle to prefer lower elevations and avoid steep slopes further from water is widely reported (Berg and Hudson 1981, Peek and Krausman 1996, Stewart *et al.* 2002). Telfer (1994) did not find definite evidence of competition between cattle and cervids; however, there was extensive overlap in use of space, habitat and forage resources overall. In a similar study, Berg and Hudson (1981) found high overlaps for temporal, topographical and habitat variables, and low dietary and spatial overlaps between mule deer and cattle. In the northwestern United States, Stewart *et al.* (2002) found strong resource partitioning between elk, mule deer and cattle as well as evidence that competition resulted in spatial displacement. Unlike cattle, elk and mule deer used similar, steeper slopes and higher elevations. There was high spatial and dietary overlap between mule deer and elk in autumn, but partitioned use of vegetation communities occurred during the summer. Deer and elk shifted habitat use to lower elevations and more level slopes when cattle were absent. In a subsequent study, the same authors found complete separation of diets between mule deer, elk and cattle in summer, despite substantial habitat overlap (Stewart *et al.* 2003). Willms *et al.* (1979) found that the potential for direct competition between mule deer and cattle on big sagebrush (*Artemisia tridentata*) range in British Columbia was greatest in spring. Both mule deer and cattle selected bluebunch wheatgrass and crested wheatgrass.

In their study of the effects of rest-rotation cattle grazing on mule deer and elk, Yeo *et al.* (1993) found that mule deer did not appreciably alter their habitat use patterns in relation to cattle grazing in mountainous terrain in Idaho. Mule deer in this study selected fields previously grazed by cattle that season and avoided rested pastures and fields with cattle present in most years. In spring, mule deer used steeper slopes in grazed fields than in rested fields, and were seen more frequently in draws in rested pastures and on slope faces in grazed fields. In winter, mule deer were observed at higher elevations and used draws more frequently and benches less frequently in rested fields than in grazed fields.

Cooper *et al*. (2008) found that while white-tailed deer and cattle in Texas often occupied the same habitat type, the two species showed strong temporal separation, rarely coming into contact with each other. Both ungulate species favoured productive habitat such as riparian areas, although white-tailed deer made greater use of rough terrain than cattle. White-tailed deer generally did not come within 50 m of cattle and tended to move away if cattle came closer.

Under appropriate management livestock grazing can benefit deer and maintain Healthy rangeland (Holechek *et al.* 1982, Peek and Krausman 1996, Teer 1996). Well-managed livestock grazing can stimulate the diversity and abundance of grass, forb and browse species that provide a variety of forage for deer throughout the seasons (Bryant *et al.* 1981). Removal of unpalatable standing dead material by fall cattle grazing, for example, benefits spring grazing opportunities for deer and elk (Short and Knight 2003). Short and Knight (2003) found that fall grazing in rough fescue (*Festuca campestris*) range in Montana increased green growth available to deer in the spring and did not compromise fescue forage production. Deer and elk select green grass growth during the spring for its high nutritive value and increased palatability (Short and Knight 2003). The benefits of fall grazing have also been demonstrated in bluebunch wheatgrass – big sagebrush range in British Columbia (Willms *et al.* 1979, 1980, 1981). Willms *et al.* (1979) noted that moderate or heavy fall grazing by cattle made spring forage more attractive to deer, while light grazing did not have an appreciable effect on deer distribution. Fall grazing benefits have not been shown for dry mixedgrass or mixedgrass prairie, where rank litter build-up is generally not a concern due to more arid conditions and less productivity. Fall grazing benefits also may not apply to valuable deer winter range due to potential competition for limited winter forage and spatial displacement of deer during critical winter periods (Short and Knight 2003).

In the absence of cattle grazing, deer foraging habitat can be reduced due to shifts in succession from communities formerly dominated by xeric shrubs, such as big sagebrush, toward a perennial grass-forb community (Austin and Urness 1998). A combination of deer and cattle grazing imposes a balance of use on all forage components and thereby prevents shifts in competitive advantage (Severson and Urness 1994). Because bison and cattle are both generalist herbivores that graze preferentially on graminoids, a similar relationship likely existed historically between bison and deer (Knapp *et al.* 1999). However, direct comparisons between bison and cattle are inappropriate given that cattle are considered more sedentary than bison and use a significantly lower percentage of upland habitat compared with bison, preferring gentler terrain and floodplain habitat (Knapp *et al.* 1999). Despite inherent differences in their grazing behaviour, cattle, as large grass-feeding herbivores, may be able to fulfill a similar ecological function as bison under appropriate management. Grazing management strategies, such as appropriate stocking rates, duration, timing and distribution tactics, can be used to achieve a greater degree of ecological equivalency between bison and cattle (Knapp *et al.* 1999).

1.6 Grazing Systems and Deer Habitat Management

Several studies have examined the effects of various grazing systems on the habitat needs and reproductive success of the mule and white-tailed deer (Reardon *et al.* 1978, Bryant *et al.* 1981, Holechek *et al.* 1982, Kie *et al.* 1991, Peek and Krausman 1996). Stocking rate and the timing and distribution of cattle are decisive factors influencing the potential benefits or detriments of each grazing system (Table III-12). Most studies indicate that deer do poorly on ranges stocked heavily with livestock that are grazed continuously year after year (Bryant *et al.* 1981). There tends to be a more positive response for deer if there is periodic resting of the range from domestic livestock (Bryant *et al.* 1981). Deer are reported to make greater use of deferred-rotation pastures, with the more frequent the deferment the higher the preference (Reardon *et al.* 1978, Holechek *et al.* 1982). A grazing system that encourages removal of mature grass followed

by periodic rest will lead to nutritious grass re-growth and therefore benefit deer during the rest period (Bryant *et al.* 1981).

The potential positive and negative implications of five grazing systems (continuous, complementary, rotational, intensive and riparian area grazing) on deer habitat are discussed further in Table III-12.

Table III-12 Grazing Systems and Deer Habitat Management

Grazing System	Discussion
Continuous (Season-Long) Grazing	
Advantages:	Provided proper utilization rates are observed, continuously grazed pastures can be managed in a manner that provides sustainable forage for both deer and cattle. Stocking rates should take into consideration the combined removal of forage biomass by cattle and deer (Telfer 1994). Light or moderate stocking rates and conservative utilization rates (25% to 50%) ensure more sustainable rangeland productivity for cattle and wild ungulates (Holechek *et al*. 1982, Kie *et al.* 1991, Peek and Krausman 1996, Teer 1996). To counter some of the potential negative impacts of continuous grazing, strategies such as salt and water placement can be used to encourage cattle away from critical deer habitat (such as riparian areas or critical winter range).
Disadvantages:	Deer have been noted to do poorly on heavily stocked ranges that are continuously grazed year after year (Bryant *et al.* 1981, Holechek *et al.* 1982, Peek and Krausman 1996). Continuous grazing at high stocking rates reduces available forage and cover for deer and fawns, resulting in potential displacement or altered activity patterns of deer, heightened predation risks and reduced productivity and survival (Kie *et al.* 1991). Availability of adequate forage during the summer is particularly crucial to adult females to enable them to meet the energetic demands of lactation (Kie *et al.*1991). Under continuous grazing, forage availability is typically reduced to a minimum by the end of the season, particularly during drought years. Over a period of years this type of grazing results in deterioration of range health. Bryant *et al.* (1981) found that diet samples from poor condition (Unhealthy) rangeland near Sonora, Texas had significantly lower yearly averages of crude protein and phosphorus than samples from higher condition (Healthy) pasture. Reduced forage quantity and quality may result in displacement of deer from native prairie and increased deer use of surrounding agricultural cropland (Swenson *et al.* 1983, Loft *et al.* 1991).

	Under continuous grazing, the greatest negative impacts from cattle grazing are often associated with unrestricted use of riparian areas. Riparian areas are particularly important to deer in the prairies for providing diverse forage and shelter (Nietfeld *et al.*1985). Heavy cattle use in riparian areas diminishes both woody and herbaceous cover through repeated utilization and trampling (Fitch and Adams 1998).
Complementary Grazing	
Advantages:	Complementary grazing, a form of rotational grazing, offers several potential advantages to deer. Complementary grazing allows deferred use of native rangeland and an extended grazing season by utilizing tame pasture in the spring. Early-season deferment of native prairie provides increased security of hiding cover for fawns and is also beneficial to improving the health of riparian areas (Fitch and Adams 1998). This system can also improve the yield and health of native prairie. Tame forage species such as meadow brome (*Bromus biebersteinii*) and Russian wild rye grass (*Psathyrostachys juncea*) initiate growth early in the season and therefore are more palatable and nutritious to both deer and cattle from late April to mid-June than are common native perennial grasses. Russian wild-rye also has a long growth season and high protein content, is drought- and cold-tolerant and is palatable in the fall (Looman 1983). Availability of high quality forage is particularly important to deer early in the season to replenish energy reserves that are depleted through the winter (Wallmo *et al.* 1977). Grass use by deer is highest in the spring (Peek and Krausman 1996). Deer favour green grasses with higher nutritive value that initiate growth early (Peek and Krausman 1996). Tame pastures, strategically placed, can therefore function to attract deer away from crop fields.
Disadvantages:	The cumulative benefits of complementary grazing depend on whether tame pastures compromise formerly productive deer habitat (*i.e.* shrubland, wetland edge habitat or important sheltering and winter habitat). If complementary grazing systems are created by seeding marginal cropland to permanent cover, the benefits of this system in terms of deer habitat are greatly increased. As deer are attracted to tame pastures in the spring, this creates the potential for competition with cattle on these pastures, particularly if there is a shortage of alternate high quality forage. Heavy use of tame pastures reduces their productivity and vigour, and means that re-seeding may be required on a more frequent basis. Hall and Stout (1999) reported that deer feeding reduced annual yield of pure alfalfa pastures by an average of 54%. Hall and Stout (1999) recommend alfalfa – grass mixtures as opposed to pure alfalfa to reduce economic losses due to deer feeding. It is possible that damage to tame pasture and cropland will be lessened if deferred grazing allows for improved native range condition. Native range typically offers a

	greater diversity of forage and cover types than tame pasture and therefore may be preferable to deer.
Rotational Grazing – Rest Rotation and Deferred Rotation	
Advantages:	Rotational grazing systems with suitable stocking rates are generally considered to be most beneficial to deer (Reardon *et al.* 1978, Holechek *et al.* 1982, Kie *et al.* 1991, Peek and Krausman 1996, Teer 1996). Grazing systems that allow for periodic rest of the range are advantageous to deer as livestock remove mature grass herbage and stimulate nutritious regrowth that is available to deer during the rest period (Bryant *et al.* 1981). Periodic rest and rotational use also allows for improved carryover and reduces selective grazing by cattle, forcing overall better utilization of the range. Rotational grazing also offers a strategy to control cattle use of riparian areas, allowing for rest or recovery of these sensitive areas and improved vigour of important shrubs and grasses used by deer. Rest-rotation systems that allow for a year of non-use can better facilitate recovery and maintenance of important browse species such as willows and red-osier dogwood (*Cornus stolonifera*) (Fitch and Adams 1998). Bryant *et al.* (1981) reported that a Merrill, 4-pasture grazing system increased the availability and use by deer of grass regrowth; supported higher densities of deer than continuous year-long grazing systems; and maintained the range in excellent condition (high range health). A Merrill grazing system is a deferred rotational grazing strategy using multiple herds. This system grazes three herds of livestock in four grazing units with one unit deferred at all times (Reardon *et al.* 1978, Holechek *et al.* 1982). Kie *et al.* (1991) stressed the importance of deferring grazing early in the season on native range to protect suitable hiding cover for deer fawns and to ensure sufficient forage for lactating females. Kie *et al*. (1991) recommend a deferred rotation grazing system with half of the allotment grazed in early summer and the other half in late summer at moderate levels, with the order of grazing rotated each year. Reardon *et al.* (1978) noted that deer preferred Merill grazing systems, with the more frequent the deferment the higher the preference. Loomis *et al.* (1991) evaluated the effects of three grazing systems on summer range in Sierra Nevada, California, on deer carrying capacities. A rest-rotation system involving one or two years of rest (non-use) for each pasture increased the potential carrying capacity of the range for mule deer in comparison to continuous moderate grazing and short-duration grazing (Loomis *et al.* 1991, Peek and Krausman 1996). Based on a synopsis of research, Austin (2000) recommended cattle grazing between May 1 and June 30 and a rest-rotation system where two-thirds of the available range is grazed annually.

Disadvantages:	Rotational grazing systems typically require higher input costs due to the need for more fencing or water developments and additional herding. These costs may be balanced by improved health of native range as well as potential for higher sustainable deer carrying capacities. Monitoring range health and adjusting stocking rates is important to ensure moderate use. As grazing is less selective under rotational grazing systems, under heavy stocking rates cattle will more heavily utilize important deer forage.
Rotational Grazing – High Intensity Grazing	
Advantages:	Intensive grazing systems with sufficient rest and frequent monitoring may be suitable for stimulating a diversity of forb and browse forage for deer by maintaining range in lower seral stages. Reardon *et al.* (1978) found that white-tailed deer density was highest in a 7-pasture short duration grazing system in which pastures were grazed for six to nine weeks then rested for the remainder of the year. However, as this study was conducted in Sonora, Texas, its applicability to southern Alberta is questionable. Few studies have shown demonstrable positive effects of intensive grazing systems for deer in southern Alberta.
Disadvantages:	High intensity, low frequency (short duration) grazing systems have been noted to have the greatest potential to damage important deer browse, forbs and seed producing species as well as cover (Teer 1996). Deer have been observed to avoid areas occupied by large numbers of cattle and shift use to other areas of their home range (Peek and Krausman 1996). This can result in deer being excluded from preferred habitat and pushed into less productive habitat with reduced cover and can lead to forage competition between deer and cattle (Kie *et al.* 1991, Loft *et al.* 1991). The implications of this to deer survival depend on the timing of grazing. Heavy use of preferred deer habitats in the spring or fall can be particularly damaging as deer have increased energy demands prior to and following winter (Kie *et al.* 1991). Heavy use in the spring can also be particularly harmful to deer and their habitat by depleting available hiding cover for newborn fawns and for causing significant damage to riparian areas at a time when banks are soft and vulnerable to compaction. Under short-duration, high intensity grazing systems, cattle are less selective and will utilize all available forage, including important deer cover or forage that would not be grazed under lower stocking rates. Intensive grazing systems can also lead to increased erosion and increased soil bulk density in mixedgrass prairie and result in loss of desirable forage species on fescue prairie (Dormaar *et al.* 1989, Willms *et al.* 1993).

Riparian Area Grazing	
Advantages:	River and creek valleys and wetland edge riparian habitat provides critical year-round deer habitat in the prairies of southern Alberta (Nietfeld *et al.* 1985). The trees, shrubs and tall herbaceous vegetation found in these habitats provide valuable shelter and security cover as well as diverse forage for deer (Nietfeld *et al.* 1985). Without appropriate management systems, cattle will linger in riparian areas, and through excessive trampling and repeated defoliation, will diminish the availability of forage and cover for deer and fawns, and cause sedimentation of the waterbody (Fitch and Adams 1998). Various riparian area grazing systems are available, such as rest- or deferred rotational grazing or corridor fencing that aim to control the timing and intensity of cattle use in riparian areas to minimize damage and facilitate rest and recovery. These types of strategies have proven to improve the health of riparian ecosystems and concurrently enhance their value as important deer habitat (Fitch and Adams 1998). Improved regeneration and long-term sustainability of woody browse, increased availability and productivity of herbaceous cover, as well as potential for improved water quality are obvious benefits to deer from implementing appropriate riparian area grazing strategies.
Disadvantages:	Even with controlled use of riparian areas by cattle, intensive deer utilization can limit regeneration and recovery of woody riparian species (Opperman and Merenlender 2000). Deer fencing or shrub or tree planting may be required in these instances to ensure long-term sustainability of certain riparian reaches. Controlling cattle use in riparian areas is also associated with economic costs due to fencing and alternate water source developments.

1.7 Range Improvements and Deer Habitat Enhancement

Aside from using cattle grazing as a habitat management tool, there are other types of range improvements that can enhance habitat for deer. Prescribed burning, for example, has been shown to increase the long-term availability, palatability and nutritive quality of forage for deer by stimulating primary productivity in subsequent years (Scotter 1980, Pearson *et al.* 1995). Pearson *et al.* (1995) monitored winter habitat use by large ungulates following a fire in northern Yellowstone National Park. Burned areas were used by ungulates more often than expected, particularly during mid- to late winter, and resulted in greater forage procurement for ungulates three to four years post-fire (Pearson *et al.* 1995). Similarly, the nutritional quality of mule deer winter diets was enhanced significantly after a fire in the Front Range of Colorado due to higher levels of crude protein in diets from burned sites (Hobbs and Spowart 1984). Mule deer obtained more green grass from burned areas (Hobbs and Spowart 1984). Fire also improved forage availability by removing dead woody tissue or litter (Hobbs and Spowart 1984, Pearson *et al.* 1995). Deer make little use of mature grass or sedges in their diet, and benefit when grass re-growth or forbs are made accessible (Collins and Urness 1983). Willms *et al.* (1980, 1981) reported that deer selected burned plants in greater proportion than grazed plants in the spring

following fall burning and grazing treatments in bluebunch wheatgrass range in British Columbia. Fire is also beneficial not only in terms of improving forage quality but also for maintaining edge habitat and preferred shrub meadow habitats (Cairns and Telfer 1980).

Other range improvements, such as seeding, may also improve deer habitat, while practices such as brush removal can negatively affect deer. Seeding marginal cropland to permanent cover of forages such as Russian wild rye (*Psathyrostachys juncea*) provides nutritious forage for deer and cattle in the early spring and late fall as part of a complementary grazing strategy (Table III-12) (Urness 1981). These grass species initiate growth earlier in the spring and produce larger amounts of fall re-growth than most native perennial grasses (Urness 1981). Brush removal negatively impacts deer by reducing cover and forage. If brush management is considered necessary to increase grass forage production for cattle, leaving brush patches is important to provide residual habitat for deer (Reardon *et al.* 1978). In this case, Teer (1996) suggests that brush is cleared in a checkerboard or strip pattern whereby 40% of the brush is left and 60% is cleared. No clearing of brush should occur in key winter ranges such as south-facing slopes (Teer 1996). Connections between brush tracts should be left intact as travel corridors (Teer 1996).

1.8 Beneficial Management Practice Recommendations

Ideal deer habitat is comprised of a diversity of plant species, a mosaic of vegetation types and an availability of varying degrees and types of cover (Teer 1996). Management of deer habitat involves managing the quality and quantity of forage as well as the types and amount of cover available to deer (Scotter 1980). It is especially important that land-use and grazing management focus on maintaining or enhancing the condition of key habitats, such as winter range, fawning sites and riparian areas (Teer 1996). The following recommendations provide a variety of means by which to enhance deer habitat in the Milk River and South Saskatchewan River Watersheds and throughout the Grassland Natural Region of Alberta (NRC 2006).

1.8.1 General Recommendations

- Maintain landscapes with a diversity of cover types, including intact riparian areas, shrublands, aspen or balsam poplar (*Populus balsamifera*) groves, shelterbelts, native prairie, tame pasture and some cropland. Habitats with greater diversity can sustain stable deer populations by increasing forage and cover opportunities throughout the year (Teer 1996).
- Maintain shelterbelts in cultivated areas to provide cover and bedding areas as well as supplemental browse for deer. Isolated shelterbelts or woody cover in open prairie is particularly important to mule deer (Nietfeld *et al*. 1985).
- Enhance, maintain and control sustainable use of riparian edge habitat around creeks, rivers and wetlands. Woody species (trees and shrubs), and productive herbaceous vegetation associated with riparian habitats, provide excellent deer forage and cover. Riparian corridors are also important as secure travel corridors for deer (Scotter 1980). Maintaining connected areas of dense cover is especially important for white-tailed deer (Nietfeld *et al.* 1985).

- Protect critical winter range habitat (south- and west-facing exposed slopes) from cultivation or development. Monitor the health of winter range forage on a regular basis (Scotter 1980).
- Avoid human disturbance in critical deer winter ranges, especially during severe winters with cold temperatures and snow depths exceeding 15 cm to 20 cm (Taggart pers. comm.).
- Minimize traffic to oil and gas wells in critical deer winter range (Sawyer *et al.* 2009).
- Maintain patches of 0.4 ha to 2.0 ha of thermal vegetation and fawning cover (trees or shrubs and tall herbaceous cover) on the landscape (Nietfeld *et al.* 1985). Maintain patches of 2.6 ha to 10.5 ha of hiding cover for deer during all seasons (Nietfeld *et al.* 1985). Maintain connectivity between patches of hiding cover, where possible (Scotter 1980).
- Maintain preferred browse species, including choke cherry, rose, snowberry, silverberry, juniper, red-osier dogwood and aspen (Nietfeld *et al.* 1985).
- Tame pastures and cropland are used for foraging by deer, particularly during early spring and fall. Alfalfa-grass mixtures may result in less yield loss due to deer feeding than pure alfalfa (Hall and Stout 1999). Where tame pastures are used as lure crops for deer to prevent impact to cropland, consider strategic placement of pastures near to adequate cover and use a forage mix with high nutritional value early and late in the season.
- Avoid converting large areas of native prairie to monoculture cropland. Cultivation creates habitat fragmentation, reduces habitat diversity and eliminates important native forage and cover (Nietfeld *et al.* 1985).
- Prescribed burning may be an appropriate tool to enhance habitat for deer in areas with thick build-up of standing dead litter, such as under-utilized rough fescue grasslands (Pearson *et al.* 1995). It is important to create a patchwork of burned and non-burned areas to provide habitat diversity and ensure sufficient forage and cover for deer in the short-term (Bryant and Morrison 1985). Leaving some areas permanently protected from burning is also important. The size and distribution of burns should be determined in consultation with local ecological experts. Burns should only be conducted in years with sufficient moisture. Bryant and Morrison (1985) recommend deferring cattle grazing in fall burned pastures between April to July. To allow sufficient rest between burns, frequency of burns should be determined according to the natural historic fire regime for the area.
- Consider the relative impact of deer on utilization of riparian woody browse. Evaluate strategies such as riparian tree or shrub planting where heavy use by deer is occurring. Upland habitat enhancement techniques can also be used to provide habitat for deer outside of riparian areas.
- Where necessary, maintain deer at sustainable population levels through the use of managed hunting to ensure sustainable use of habitats.

1.8.2 Grazing Recommendations

Grazing can be used as a tool to enhance deer habitat by increasing plant species diversity, reducing build-up of thick, standing-dead litter and encouraging growth of early-season forbs

and grasses (Peek and Krausman 1996). Grazing systems that will benefit deer need to maintain suitable stocking rates that allow for maintained grassland productivity and consider the combined removal of forage biomass by both livestock and deer. In general, competition between domestic livestock and deer is most severe on degraded ranges that are heavily stocked (Teer 1996). Controlling the timing, intensity and frequency of grazing impacts the degree of potential competition between cattle and deer. Grazing management decisions need to be tailored to meet local ecological conditions, taking into consideration how changes in management will impact important deer forage and cover through the seasons. Adequate winter range forage, availability of early-season grasses and adequate hiding cover for fawns are especially important considerations. Appropriate grazing systems for deer should be developed site-specifically for ranches participating in conservation programs in southern Alberta.

The following list summarizes key livestock grazing management principles where the goal is to enhance deer habitat:

- Use livestock grazing in combination with deer herbivory to maintain a balanced grass/forb/browse ratio (Scotter 1980).
- Maintain native grassland utilization rates of 25% to 50% to ensure a sufficient degree of carry-over remains and rangeland productivity is sustained (Adams *et al.* 2005, 2013a, b).
- Monitor utilization rates of woody browse and determine the use-tolerance of preferred deer browse species that does not result in lost vigour or productivity (Scotter 1980). Manage for sustainable cattle and deer browse use through the use of fencing, rest and light to moderate stocking rates.
- Track annual grazing capacities and vary stocking rates and distribution in accordance with drought events and range and riparian health. Consider the contribution of deer herbivory to overall forage utilization when setting stocking rates.
- Avoid intensive grazing systems with high density stocking rates over large areas.
- Use complementary grazing as a means to defer early-season use of native prairie and provide supplemental tame forage for deer in the spring and fall.
- Implement rotational grazing systems to provide parts of the range with deferred early-season grazing and controlled periods of rest (Reardon *et al.* 1978, Holechek *et al.* 1982, Kie *et al.* 1991, Peek and Krausman 1996, Teer 1996). Deferred use and rest benefits the availability and quality of deer forage and improves security of hiding, thermal and fawning cover. Rotational grazing systems also provide a tool to manage cattle use of critical riparian area habitat (Fitch and Adams 1998). One season of rest is sufficient to improve herbaceous cover, while a longer period of rest may be required to improve woody species cover (Scotter 1980).
- Where appropriate, use fall grazing to improve spring grass and forb forage availability and quality for deer in fescue prairie (Peek and Krausman 1996, Short and Knight 2003). Livestock grazing can be used to improve vegetation conditions for deer by removing old growth and stimulating the production of new growth to provide palatable deer forage in the spring (Peek and Krausman 1996).

- Avoid over-utilization on critical winter ranges, including south- and west-facing slopes along valleys or coulees and treed or densely-vegetated riparian corridors. Avoiding late-season use of these areas is important to ensure sufficient forage remains for deer during the winter months (Short and Knight 2003). Where necessary, fencing can be used to restrict cattle access to key winter-range habitat.
- Manage and monitor cattle use of upland woody vegetation to ensure trees and shrubs are healthy and capable of regenerating. Avoid placing salt near forested or shrubby draws.
- Manage livestock grazing in riparian areas to enhance habitat value (forage and cover) for deer (Collins and Urness 1983).
- The optimal time of use of riparian areas is during the summer after spring runoff when the streambanks are no longer soft, and before the dormant season (Fitch and Adams 1998).
- Distribute salt away from riparian habitats to reduce impact to these areas and encourage better utilization of the range.
- Develop upland stock water, where necessary, to control heavy use of riparian areas. Stock water provides alternate watering sites for cattle as well as deer. Availability of alternate watering sites improves deer distribution in dry years, in particular (Scotter 1980). Nietfeld *et al.* (1985) stated that optimum mule deer habitat has open water within 0.8 km of any point. Only water sources in proximity to sufficient cover are likely to be used by white-tailed deer (Nietfeld *et al.* 1985).
- Monitor use of supplemental feeding sites by deer. Because white-tailed deer are attracted to feeders, supplemental feeding in fixed locations has the potential to lead to over-use of preferred browse near feeders (Cooper *et al.* 2006). Feeder locations should be moved periodically to prevent range degradation.
- Deferred-rotation, rest-rotation, riparian pasture and corridor fencing have been suggested as techniques for improving the health of riparian areas (Fitch and Adams 1998). Several years of rest may be required where the goal is to regenerate trees like cottonwoods (Fitch and Adams 1998).

1.9 Research Recommendations

To refine specific cattle grazing recommendations for mule and white-tailed deer in the Milk River and South Saskatchewan River Watersheds, additional local information regarding deer habitat requirements, deer-cattle interactions and deer effects on agricultural resources is required. Additional information regarding various deer range improvement techniques (*e.g.*, prescribed burning) would also be valuable.

Habitat suitability modeling would provide a valuable tool for defining and mapping out the amounts and kinds of habitats frequently used by deer within the Milk River and South Saskatchewan River Watersheds and rating the importance of each habitat type. Development of Habitat Suitability Index (HSI) models for deer should be based on existing habitat use and population survey information. Traditional ecological knowledge acquired through landowner and hunter consultation can be valuable in confirming local deer use patterns. Where necessary, habitat selection studies could be used to confirm or analyze deer use patterns throughout the

year. Final models should aim to map out the following features within the Milk River and South Saskatchewan River watersheds:

- Key winter ranges
- Critical thermal, hiding, escape or fawning cover
- Prime spring and summer ranges.

Research concerning the interaction between livestock and deer should consider an evaluation of the following factors:

- Relative use by deer of habitats with varying intensities or frequencies of cattle grazing.
- The potential for cattle and deer competition or resource partitioning at different times of the year.
- The relative importance of forage versus structural features to deer and how each is affected by cattle grazing.
- The effects of cattle grazing on the quality and sustainability of critical deer winter range habitat.
- Application of fall grazing to enhance deer spring forage.
- Relative cattle and deer utilization of critical riparian habitats.
- A comparative evaluation of various grazing strategies to enhance deer habitat and minimize impact to agricultural resources.

Planned studies with experimental manipulation of livestock to control timing and intensity of grazing will be beneficial in evaluating the response of deer to various cattle grazing strategies (Teer 1996, Stewart *et al.* 2002). As deer use patterns are affected by cattle grazing, assessing preferential deer habitats requires manipulation of grazing levels as well as studying control areas that are ungrazed by cattle (Stewart *et al.* 2002). As human use patterns also have an obvious impact on deer behavioural patterns, studies should also evaluate habitat use relative to varying degrees of human activity (Scotter 1980).

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III: REPTILES AND AMPHIBIANS

A. HERPETILE GROUP 1 (PRAIRIE RATTLESNAKE, BULLSNAKE, SHORT-HORNED LIZARD AND EASTERN YELLOW-BELLIED RACER)

1 INTRODUCTION

The purpose of this report is to summarize and compare the ecological and habitat requirements of four reptile species found within the Milk River and South Saskatchewan River watersheds: the prairie rattlesnake (*Crotalus viridis*), bullsnake (*Pituophis catenifer sayi*), short-horned lizard (*Phrynosoma hernandesi hernandesi*), and eastern yellow-bellied racer (*Coluber constrictor flaviventris*). Based on this information, the potential effects of grazing and various grazing systems on these reptiles, their habitat and their respective prey are discussed. This discussion is followed by a summary of recommended beneficial management practices to enhance or protect reptile populations and their habitat throughout their range in southeastern Alberta. Lastly, a brief summary of additional information needs is presented.

Prairie rattlesnakes, bullsnakes, short-horned lizards and eastern yellow-bellied racers are at or near the northern edges of their North American ranges in Alberta, and are very locally distributed in southeastern Alberta. For example, the majority of the breeding range of the short-horned lizard in Alberta is found within the Milk River Watershed. To enable them to survive the harsh winters of a northern climate, these reptiles hibernate in overwintering sites known as hibernacula. Other key habitat components include birthing areas (also known as rookeries) (rattlesnakes), nesting sites (bullsnakes) and productive foraging areas. Habitats that provide opportunities of thermoregulation are key determinants in the selection of breeding, wintering and foraging habitat by these reptile species. Due to their limited range or rarity in the Canadian prairies, suspected population declines and their sensitivity to human disturbances and extreme weather events, their populations are considered Sensitive or At Risk in Alberta (with the exception of the racer, which is considered Accidental/Vagrant) (Government of Alberta [GoA] 2016). Because cattle grazing is the dominant land-use within the ranges of these species in the Milk River and South Saskatchewan River watersheds, understanding the potential impacts of grazing on these species and their habitats is important for their management and long-term conservation. It is also important to consider strategies to minimize potential impacts due to other human activities, such as industrial development, which can pose a risk to these species or their habitats.

2 PRAIRIE RATTLESNAKE

2.1 Background

Prairie rattlesnakes are tan in colour with dark-coloured bands or blotches along its back and olive-or-brown coloured tail rings. The prairie rattlesnake was previously thought to be a subspecies of the western rattlesnake (*Crotalus oreganus*), but is now considered a distinct species (*Crotalus viridis*) (Crother *et al*. 2012). The range of this species in North America extends from northern Mexico through the central United States and into southeastern Alberta and southern Saskatchewan (Watson and Russell 1997). The prairie rattlesnake is found

primarily along major river watersheds in southeastern Alberta, with most hibernacula located along the breaks and coulees of the South Saskatchewan, Red Deer, Bow, Oldman and Milk Rivers (Gannon 1978, Kissner and Nicholson 2003a).

The prairie rattlesnake has a General Status listing of 'Sensitive' in Alberta (GoA 2016). The species is currently listed as a species of 'Special Concern' under the provincial *Wildlife Act* (GoA 2014) as well as nationally by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC 2016). *Crotalus viridis* is listed as species of 'Least Concern' by the International Union for Conservation of Nature and Natural Resources (IUCN) (Frost *et al.* 2007). The prairie rattlesnake is designated as a 'non-game animal' under Alberta's *Wildlife Act*, making it illegal to kill, possess, buy or sell rattlesnakes. Prairie rattlesnake hibernacula are afforded year-round protection from destruction or human disturbance under the *Wildlife Act*.

The most recent population estimate for Alberta was 12,672 mature individuals at 192 hibernacula (Alberta Environment and Sustainable Resource Development [ESRD] and Alberta Conservation Association [ACA] 2012). Due to incomplete information, this estimate was based on an assumed number of active hibernacula and the average number of mature individuals that are typically found in hibernacula. Although historical data is limited, anecdotal reports suggest that the Alberta prairie rattlesnake population appears to have undergone a long-term decline in distribution and abundance since European settlement (Watson and Russell 1997, Kissner and Nicholson 2003b; Proctor *et al.* 2009, ESRD and ACA 2012). However, it has not been possible to accurately analyze prairie rattlesnake population trends or to quantify suspected declines due to a lack of long-term studies on prairie rattlesnakes in Alberta. Although the species' range appears to have remained stable over the last half century, there is evidence that some hibernacula in Alberta have seen reductions in snake numbers since 1990 (Proctor *et al.* 2009, ESRD and ACA 2012). Only one prairie rattlesnake population, located in Dinosaur Provincial Park, appears to have rebounded from past declines (Martinson 2009a, cited in ESRD and ACA 2012). Further monitoring of prairie rattlesnake populations is required to conclusively determine trends.

The distribution of prairie rattlesnake populations in Alberta is naturally limited by climatic factors and the availability of suitable hibernacula, summer foraging areas and appropriate birthing areas (Watson and Russell 1997). Declines in prairie rattlesnake abundance and distribution are thought to be a result of habitat fragmentation, loss of native prairie habitat and direct mortality (Watson and Russell 1997, Ernst 2002). Intensive agricultural activities, human persecution, den site vandalism/destruction, urbanization, roadway traffic and energy resource development can all negatively impact prairie rattlesnake populations (Watson and Russell 1997, Ernst 2002, Martinson 2009b, Proctor *et al.* 2009). Prairie rattlesnakes are also vulnerable because numerous snakes overwinter in the same hibernacula each year. Recovery of prairie rattlesnake populations is slow because of naturally slow growth rates (ESRD and ACA 2012).

2.2 Ecology

Prairie rattlesnake distribution patterns, movements and behaviour are strongly influenced by age, sex and reproductive state (Gannon and Secoy 1984). Prairie rattlesnakes generally emerge from hibernacula in late April or early May and return in mid-September (Duvall *et al.* 1985,

Gannon and Secoy 1985, Ernst 2002). Prairie rattlesnakes at more northern latitudes such as in Alberta experience lower body temperatures during hibernation and have a shortened active season than more southerly conspecifics in the United States (Watson and Russell 1997). Early fall frosts and shorter days encourage rattlesnakes to move back to dens. All age-classes and sex classes may occupy the same den (Gannon and Secoy 1984). In the spring, males and non-gravid (non-pregnant) females migrate into the surrounding prairie to forage and mate, whereas gravid (pregnant) females remain near the denning complex (Gannon and Secoy 1984).

Most snakes (70%) are oviparous and lay eggs with soft leather shells that protect their developing young (Bartlett and Tennant 2000). Prairie rattlesnakes, however, are ovoviviparous, with females giving birth to live young. Although some female rattlesnakes in Saskatchewan have been found to occasionally give birth annually (Kissner pers. comm.), the majority of female prairie rattlesnake reproductive cycles are biennial, triennial or longer, meaning that individual females produce offspring at most every two years (Gannon and Secoy 1985). Breeding takes place in the summer or early fall and young are born the following year from late August to mid-October (Watson and Russell 1997, Ernst 2002). Breeding occurs on summer ranges or at or near hibernacula (Kissner pers. comm.). Pregnant females usually give birth to four to 13 young (Jørgensen and Nicholson 2007) near the hibernacula at birthing areas (rookeries) thought to be chosen for their favorable microclimates, and possibly for factors that decrease their risk of predation (*e.g.*, cover). Male snakes attain sexual maturity at three to four years of age, while female snakes mature at five to seven years of age. Given their late reproductive maturity, relatively small litter size, and biennial or greater reproductive cycles, prairie rattlesnakes have a slow population growth rate (Watson and Russell 1997).

2.2.1 Diet

Upon emerging from hibernation, snakes move to upland habitats in search of prey, often remaining in areas with high small mammal abundance (Didiuk 1999).

Prairie rattlesnakes prey on small mammals, birds, amphibians and other reptiles (Russell and Bauer 2000) and locate their prey both visually and by using two heat-sensing pits on their upper jaw (Ernst 2002). The exact composition of the prairie rattlesnake diet varies over its range in relation to the abundance and composition of the local prey base (Hill *et al.* 2001). Hill *et al.* (2001) examined the gut contents of 20 road-killed prairie rattlesnakes and the composition of 8 scats from wild-caught individuals in a multiple land use area in southeastern Alberta. Sagebrush voles (*Lagurus curtatus*) were the dominant prey consumed, constituting 53% and 68%, respectively, of prey items in gut contents and scats. Meadow voles (*Microtus pennsylvanicus*) were the next most frequently consumed prey, constituting 38% of gut contents and 8% of scats. Olive-backed pocket mice (*Peragnathus maniculatus*), western jumping mice (*Zapus princes*) and Richardson's ground squirrels (*Spermophilus richardsonii*) made up a minor portion of the prairie rattlesnake diet. The low occurrence of Richardson's ground squirrels in prairie rattlesnake diets may have been due to their apparent low density within the study area (Hill *et al.* 2001). Prairie rattlesnakes were found to consume multiple prey items of the same species, indicating that snakes tend to exploit patches with an abundance of colonial burrowing prey species (Hill *et al.* 2001). Other studies in Alberta have found that northern pocket gophers (*Thomomys talpoides*) are an important component of prairie rattlesnake diets (Didiuk 1999).

2.2.2 Predators

Common predators of prairie rattlesnakes include American badgers (*Taxidea taxus*), coyotes (*Canis latrans*) and various raptor species (Watson and Russell 1997).

2.3 Habitat Requirements

2.3.1 General

In Alberta, the prairie rattlesnake is found within the Mixedgrass and Dry Mixedgrass Subregions of the Grassland Natural Region (Natural Regions Committee [NRC] 2006), most commonly along river valleys and associated coulees, badlands and sage flats (Watson and Russell 1997).

Key habitat components for prairie rattlesnakes include hibernacula (wintering habitat), foraging and thermal cover habitat (summer habitat), and birthing (rookery) sites. Habitat Suitability Index (HSI) models were developed for each of these habitat components for prairie rattlesnakes in the Milk River Basin (Kissner 2004). The wintering habitat HSI model included three, equally weighted variables: distance from escarpment of major river, drainage or coulee; aspect; and a landscape model variable from the Agricultural Region of Alberta Soil Inventory Database (AGRASID). According to this model, sites that are most likely to offer suitable prairie rattlesnake hibernacula occur within 4 km of a major river, coulee or drainage, on south-, east-, or southeast-facing slopes, and are in areas with rough terrain (high relief, moderate to steep slopes and greater than 10% exposed bedrock). Hibernation sites are also known to occur in areas of slumping; however, geographic information system (GIS) coverage was not available for this variable.

The summer habitat HSI model was more complex, and included five variables: distance from major river, drainage or coulee; road density; road type; native prairie class; and shrub or tree cover (Kissner 2004). According to this model, optimal prairie rattlesnake summer habitat occurs within an area of 25 km of a major river, coulee, or drainage, that has low densities of roads (in particular highways and secondary highways) and is characterized by native prairie cover with low to moderate densities of shrubs. Areas with high human densities are also considered unsuitable as summer prairie rattlesnake habitats.

The HSI model for identifying potential rookery habitat included four variables: distance from major river, drainage or coulee; aspect; shrub cover; and bare rock cover (Kissner 2004). According to this model, optimal rookery habitat occurs in areas that are within 1 km to 5 km from a major river, coulee and drainage, and that have south, east, or southeast aspects with moderate shrub cover or bare rock cover. The overall HSI equation included elevation as a determining variable. In general, prairie rattlesnakes do not occur above 1200 m in the Milk River Basin of Alberta (Kissner 2004).

2.3.2 Hibernacula

Prairie rattlesnake occurrence in the Milk River and South Saskatchewan River watersheds is strongly influenced by the availability of suitable hibernacula (overwintering dens). The protection of denning sites is considered critical for the conservation of prairie rattlesnakes

(Nicholson and Rose 2001). Without adequate hibernacula sites, prairie rattlesnakes are unable to survive the long, cold winters of northern climates (Watson and Russell 1997). Hibernacula generally occur along river escarpments, often in stable slump blocks, meander scarps and fissures, subterranean water channels, sinkholes and rocky outcrops (Gannon 1978, Watson and Russell 1997, Didiuk 1999). Burrowing animals such as American badgers play an important role in creating and maintaining hibernacula sites (Didiuk 1999, Fast and Gates 2003). Didiuk (1999) reported that 8 of 11 den sites on the Canadian Forces Base (C.F.B.) Suffield were coyote or badger burrows.

An underground hole, crevice, mammal burrow or other retreat used as hibernacula generally must be deep and extend to a depth below the frost line (Duvall *et al.* 1985). Hibernacula are most often located on south, east or southeast-facing slopes as these aspects provide maximum solar insolation and protection from prevailing winds (Watson and Russell 1997, Didiuk 1999). Hibernacula may occur on other aspects, even north-facing slopes, provided that these sites have suitable (warm) basking sites (Didiuk 1999, Andrus 2010).

Prairie rattlesnakes tend to return to the same hibernacula year after year, provided the den is not disturbed or destroyed (Watson and Russell 1997, Didiuk 1999, Jørgensen 2009, Andrus 2010). Consequently, hibernacula may be used by many generations of snakes. Scent trails or pheromones are used to identify past travels and locate denning sites (Watson and Russell 1997). Prairie rattlesnakes have been known to share hibernacula with bullsnakes, wandering garter snakes (*Thamnophis elegans*), plains garter snakes (*T. radix*), and occasionally plains hognose snakes (*Heterodon nasicus*) and eastern yellow-bellied racers (Watson and Russell 1997, Kissner 2004, Kissner *et al.* 1996, Wright and Didiuk 1998, Gardiner *et al.* 2013). Because large numbers of snakes often den together, the destruction of a single hibernaculum may have a severe impact on snake populations (Kissner 2004).

2.3.3 Gravid Female and Birthing (Rookery) Habitat

Gravid female habitat is strongly associated with a suitable birthing area or rookery (Watson and Russell 1997) Gravid females aggregate at birthing rookeries that are usually close to the denning area, and females seldom move away from these sites (Gannon and Secoy 1985, Watson and Russell 1997). Rookeries are often characterized by the presence of large, flat table rocks overlaying abandoned mammal burrows (Duvall *et al.* 1985). These areas also tend to be associated with greater proportions of sand and shrubs (*i.e.*, silver sagebrush [*Artemisia cana*]) than the surrounding prairie (Fast and Gates 2003). Southeast aspects and presence of burrows (particularly in the absence of rock outcrops or rock piles) are other critical components of gravid female habitat (Fast and Gates 2003). Burrows provide critical shade and shelter from predators and aid in thermoregulation (Graves and Duvall 1993). Rookeries are often used in consecutive years concurrently by several different females (Gannon and Secoy 1985). These females bask in large groups throughout the summer months, and may not feed the year they give birth (Graves and Duvall 1993). Once thought to be sedentary, it is now believed that gravid females make short distance movements around and between rookeries (Jørgensen and Nicholson 2007). Following their birth, neonates (baby snakes) remain in the rookery area until they migrate to hibernacula in September. Females do not provide any parental care to their offspring (Ernst 2002).

2.3.4 Foraging Habitat

Suitable foraging sites tend to be located in native prairie with sagebrush and mammal burrows to provide cover for thermoregulation, escape from predators and prey ambush locations (Gannon and Secoy 1985, Duvall *et al.* 1985). Burrows are not only used as refugia during periods of inactivity, but may also be the centre around which feeding behaviour is organized (Chizar and Cameron 1996). Prairie rattlesnakes are ambush predators and often occupy burrows waiting for rodents to enter. Due to a relatively short active season for snakes in Alberta, suitable foraging sites need to be available within reasonable distances from hibernacula to allow snakes sufficient time to accumulate reserves for winter hibernation (Watson and Russell 1997). Didiuk (1999) found that most prairie rattlesnakes in C.F.B. Suffield dispersed into the Middle Sandhills and upland prairie habitats from hibernacula along river valleys. Prairie rattlesnakes were also frequently captured in grass terraces along river meanders (Didiuk 1999). Intensive agriculture and rodent control programs reduce the availability of suitable foraging habitat and reduce prey availability for prairie rattlesnakes (Watson and Russell 1997).

Resource selection functions (RSFs) were developed for prairie rattlesnake use of summer habitat in Grasslands National Park and surrounding areas (Gardiner *et al.* 2015). Habitat variables were measured on a site-specific (*i.e.*, <1 m) basis. The model with the best fit included five variables: distance to the nearest burrow; percent bare ground cover; percent shrub cover; vegetation density (measured using a Robel pole); and maximum vegetation height. Despite being included in the model, the latter two variables were not important predictors of microhabitat use by prairie rattlesnakes. Prairie rattlesnakes selected for close proximity to burrows and areas with higher shrub cover and lower bare ground cover. None of the variables were important predictors of habitat use at the local level (*i.e.*, <10 m), suggesting that prairie rattlesnakes select for habitat at a very fine scale (Gardiner *et al.* 2015). Shrubs and burrows likely provide important retreat sites for prairie rattlesnakes where they may find prey, protection from predators, suitable reproduction spots and important thermoregulation habitat (Gardiner *et al.* 2015).

RSFs were also developed for prairie rattlesnakes in Alberta as part of the MULTISAR project (Martinson and Weilki 2012). According to the best-fit model, prairie rattlesnakes in southern Alberta select for steeper slopes, sites with higher solar radiance, and habitats closer to slopewash, native/natural ecological sites, Thin Breaks ecological sites, bedrock and rivers. Native/natural ecological sites include native upland ecological sites classified according to the Grassland Vegetation Inventory (Site IDs 11 to 24) (Alberta Sustainable Resource Development [SRD] 2010). Thin Breaks ecological sites typically occur on moderate to steep slopes with bedrock within 1 m of the soil surface and are intermediate in physical and vegetational characteristics between Limy and Badlands/Bedrock ecological sites (SRD 2010). Slopewash refers to deposits of soil and rock transported down slopes by runoff.

Research suggests that prairie rattlesnakes undertake resource partitioning with other snake species, including bullsnakes and the eastern yellow-bellied racers in southern Saskatchewan (Gardiner *et al.* 2013), as well as the Great Basin gophersnake (*Pituophis catenifer deserticola*) in Idaho (Diller and Wallace 1996). On their summer range in Saskatchewan, prairie rattlesnakes select for black-tailed prairie dog (*Cynomys ludovicianus*) colonies, hills/slopes and riparian

areas (Gardiner *et al.* 2013), whereas roads appear to be used by prairie rattlesnakes according to their availability in the environment (*i.e.*, no preference is shown). Bullsnakes and racers have some overlap in their habitat preferences with prairie rattlesnakes (primarily hills/slopes), but their highest preferences are for other macro-habitat types (Gardiner *et al.* 2013).

2.3.5 Thermal Cover

Snakes are ectothermic ("cold-blooded") animals meaning that their body temperatures fluctuate with the thermal environment (Russell and Bauer 2000). Most snakes cannot survive exposure to direct sunlight with temperatures over 38 degrees Celsius, or prolonged periods of below freezing temperatures (Duvall *et al.* 1985). Lethal temperatures for snakes depend on the time of exposure. High or low temperatures cause prairie rattlesnakes to seek escape cover or shady areas. Prairie rattlesnakes commonly use burrows, rock piles, bases of sagebrush or dense grass for shade or thermal cover (Didiuk 1999). Treed areas such as cottonwood stands may also be used for thermal relief and cover (Ernst 2002). Road surfaces may provide basking opportunities, increasing the susceptibility of prairie rattlesnakes to road mortality (Kissner 2004).

2.3.6 Area Requirements

Adult males and non-gravid females may undertake long migrations into the surrounding valley and upland areas to forage and mate (Gannon and Secoy 1985). Despite most hibernacula occurring along major river valleys in southern Alberta, and despite the fact that prairie rattlesnakes can swim well, most individuals do not cross rivers and instead remain on the same side of the river as their hibernacula (Jørgensen 2009, Andrus 2010). While prairie rattlesnakes are known to travel more than 20 km, the majority of snakes tend to be found within 15 km of their denning area (river valley or associated coulee) (Nicholson pers. comm.). At the Suffield National Wildlife Area, radio-tagged males and post-gravid females traveled up to 24.7 km from their hibernacula (Didiuk 1999). Jørgensen (2009) observed migration distances of 0.3 km to 12 km for prairie rattlesnakes in southern Alberta. Migration paths to and away from hibernacula sites are generally similar (Didiuk 1999). An exception to this general rule is prairie rattlesnakes migrating through urban environments (Andrus 2010).

Prairie rattlesnake home range sizes and movement patterns were recently determined for southern Saskatchewan populations using radio telemetry and GIS (Gardiner *et al.* 2013). Prairie rattlesnakes there had average home range sizes of 109 ha. Home range size was not significantly different between males and females. Home ranges were 'dumbbell-shaped', with areas of use concentrated around the hibernacula and summer range, which were connected by a narrow movement corridor. Distances travelled from denning sites ranged from 0.5 km to 11 km, with an average daily movement rate of 92 m. Prairie rattlesnakes in this study showed two different movement patterns: (1) those that remain in close proximity to the den and (2) those that disperse great distances from the den. Movement patterns did not correspond with sex. Jorgensen (2009) also observed short- and long-distance migrants, an observation he attributed to habitat type, with the former occupying riparian areas and the latter migrating to upland habitat.

3 BULLSNAKE

3.1 Background

The bullsnake is a large yellow- or cream-colored snakes with dark spots and a dark line across the face. Bullsnakes are restricted in occurrence to southern Alberta and southwestern Saskatchewan (Kissner and Nicholson 2003b). The range of this species extends south to northeastern Mexico and west from the Idaho panhandle to western Indiana (Russell and Bauer 2000). In Alberta, bullsnakes occur within the Grassland Natural Region, primarily in the Dry Mixedgrass and Mixedgrass natural subregions (Kissner and Nicholson 2003b; NRC 2006). In Alberta, historical records indicate that bullsnakes ranged from the Canada-United States border as far north as Trochu, and east of Calgary to the Alberta – Saskatchewan border (Kissner and Nicholson 2003b). Bullsnakes, like prairie rattlesnakes, are associated with major river valleys such as the South Saskatchewan, Red Deer, Bow, Oldman and Milk rivers (Kissner and Nicholson 2003b). Bullsnakes can reach over 2 m in length, making it the second largest snake species in Canada (Allen 1996). Bullsnakes are commonly misidentified as prairie rattlesnakes due to their similar appearance and habitat use.

The bullsnake is listed as a 'Sensitive' species in Alberta under the General Status listing (GoA 2016). The bullsnake is listed as a 'non-license' species under the provincial *Wildlife Act*, meaning there are no hunting or harvest restrictions for the species. However, COSEWIC designated the bullsnake as 'Data Deficient' in Canada in 2002, a designation the species currently retains (COSEWIC 2016). The bullsnake is listed as a species of 'Least Concern' by the IUCN (Hammerson 2007).

There is a lack of scientific information about bullsnake population sizes and trends in Alberta and Saskatchewan (Kissner and Nicholson 2003b, COSEWIC 2002). Currently, populations are thought to be stable or possibly declining (GoA 2016).

There are several limiting factors affecting bullsnake populations in Alberta, including natural predation and human persecution. The largest threats to bullsnake populations are thought to be habitat loss and alteration due to agricultural activities, industrial development and urbanization (Kissner and Nicholson 2003b). As with prairie rattlesnakes, road mortality is also a serious limiting factor for bullsnakes (GoA 2016). For example, Didiuk (1999) reported that 101 of 249 snakes (41%) were found dead on roads over a two-year period at C.F.B. Suffield. Road mortality is particularly severe on roads that occur close to hibernation sites along river valleys. Direct human persecution is also a perceived threat to bullsnake populations in Alberta (COSEWIC 2002, GoA 2016).

3.2 Ecology

In Alberta, bullsnakes typically emerge from hibernacula between late April and mid-June and return to overwintering sites in late August to mid-October (Kissner and Nicholson 2003b). Mating often occurs near the hibernacula in the month of May, after which females move to suitable nesting sites. Bullsnake reproduction varies by geographic location, but generally occurs every one to two years once mature (COSEWIC 2002). For example, most (97%) bullsnakes in southwestern Idaho were found to reproduce annually (Diller and Wallace 1996), whereas only

64% to 95% of bullsnakes in the Nebraska Sandhills reproduced in a given year (Iverson *et al.* 2012). The proportion of females reproducing in a given year is a function of body size and warmth of the previous summer, with larger females producing larger clutches and females producing larger clutches in years with warmer April-May temperatures (Iverson *et al.* 2012). Bullsnake reproduction is also a function of sexual maturity, with male bullsnakes reaching sexual maturity at one to two years of age, and females reaching sexual maturity at three to five years of age (COSEWIC 2002)

Like most snakes (70%), bullsnakes are oviparous and lay eggs with soft leather shells (Bartlett and Tennant 2000). Clutch size averaged 9.5 in the Sandhills of Nebraska (Iverson *et al.* 2012), 6.9 in Idaho (Diller and Wallace 1996) and 4.6 in British Columbia (Shewchuk 1997, cited in COSEWIC 2002). Clutch size is dependent on adult female body size and may be greater with higher spring (April to May) temperatures (Iverson *et al.* 2012). Females do not incubate their eggs and leave the nesting site shortly after their eggs are laid. In Alberta, neonatal snakes hatch in mid-August to mid-September, following a 50 to 60 day incubation period (Kissner and Nicholson 2003b). The rate of development of the embryo is temperature-dependent due to the ectothermic nature of snakes. Bullsnake hatchling size, adult size and juvenile growth rate appear to decrease with increasing latitude (Iverson *et al.* 2008). Male bullsnakes mature at one to two years of age, whereas females reach sexual maturity at three to five years of age (COSEWIC 2002).

3.2.1 Diet

Bullsnakes are considered generalist feeders and actively search for prey under rocks, in burrows, around vegetation and on perching sites (Diller and Wallace 1996, Rodriguez-Robles 2002). Their diet includes small mammals (*e.g.*, rabbits, northern pocket gophers, ground squirrels, weasels, mice, and voles), birds, bird eggs, other reptiles, and arthropods (Russell and Bauer 2000, Rodriguez-Robles 2002). Their diet is very similar to that of prairie rattlesnakes, which are considered competitors in some locations (Jackson 1945). Unlike prairie rattlesnakes, however, bullsnakes are capable tree climbers, and are regularly observed in trees, likely in search of birds, eggs and nestlings (Kissner and Nicholson 2003b). Bullsnakes kill larger prey items by constriction, whereas smaller prey are swallowed alive. Bullsnakes may feed fewer than 20 times during any particular active season (Sterner *et al.* 2002).

Bullsnake diet varies by geographic location and depends largely on what is available in the environment (Rodriguez-Robles 2002). Based on observations at C.F.B. Suffield, bullsnakes migrate to areas of high prey abundance and remain there until the onset of hibernation (Didiuk 1999). At C.F.B. Suffield, snakes migrated to northern pocket gopher burrow complexes. In Idaho, the diet of the related subspecies Great Basin gophersnake (*Pituophis catenifer deserticola*) included primarily mountain cottontails (*Sylvilagus nuttallii*) (34% of prey biomass), Townsend ground squirrels (*Spermophilus* townsendii) (30% of prey biomass) and deer mice (*Peromyscus maniculatus*) (16% of prey biomass) among other prey species (*e.g.*, voles and other mice species) (Diller and Wallace 1996). In northern Utah, bullsnakes feed mainly on mice and voles (Parker and Brown 1980, cited in COSEWIC 2002).

3.2.2 Predators

In Alberta, the bullsnake is preyed upon by a variety of animals, including hawks, badgers, foxes (*Vulpes* spp.), coyotes and striped skunks (*Mephitis mephitis*) (Russell and Bauer 2000, Kissner and Nicholson 2003b). Humans can also play a large role in bullsnake mortality, with road traffic and agricultural activities being the predominant causes (Kapfer *et al.* 2008a). Predation on nests can have a significant impact on recruitment in bullsnake populations because eggs are left untended in nests, and several females may lay their eggs in the same nest (Kissner and Nicholson 2003b).

3.3 Habitat Requirements

3.3.1 General

Bullsnakes occur in short and mixedgrass prairie habitats particularly in sandy and brushy areas and in badlands and rocky outcrops along major river valleys (Kissner and Nicholson 2003b). The species can also be found in trees in riparian forests (Kissner and Nicholson 2003b). Didiuk (1999) found that bullsnakes at C.F.B. Suffield were predominantly located in grass terrace and grass moraine habitats. Snakes were also frequently captured near old buildings and at the interface of plains cottonwood (*Populus deltoides*) and silver sagebrush habitats along river valleys.

3.3.2 Hibernacula

Bullsnakes commonly use the same types of hibernacula as prairie rattlesnakes including slump blocks, meander fissures and scarps, sinkholes, rocky outcrops and mammal burrows along major river valleys (Kissner and Nicholson 2003b). Bullsnakes often den with prairie rattlesnakes, but will den in conspecific groups in areas where other snake species are not common. In Alberta, most known bullsnake hibernation sites are located along breaks and coulees of the South Saskatchewan, Red Deer, Bow, Oldman and Milk Rivers (Kissner and Nicholson 2003b). Similar to prairie rattlesnakes, bullsnakes select hibernacula on south-, east-, or southeast-facing slopes as these aspects provide protection from prevailing winds and offer maximum solar exposure (Kissner and Nicholson 2003b).

3.3.3 Gravid Female and Nesting Habitat

Because female bullsnakes do not incubate their eggs they must find nesting sites that will provide adequate incubation conditions. Females will excavate nesting sites in sandy or friable soils and often excavate existing burrows of mammals, bank swallows or other bullsnakes. Nesting sites, like hibernation sites, are often located on south, east, or southeast aspects to provide favourable thermal conditions for embryonic development (Kissner and Nicholson 2003b). Nesting sites have been found in Alberta along sandy coulee slopes and slumps along the South Saskatchewan and Red Deer Rivers. Several of these sites are located near hibernation sites. As many as two dozen females were observed at a nesting area in southern Alberta (J. Wright, unpubl. data). Communal nesting behaviour is not uncommon among bullsnakes perhaps indicating that prime nesting habitat is limited or that there is a benefit for females to lay their eggs together (Kissner and Nicholson 2003b).

3.3.4 Foraging Habitat

In Alberta, bullsnakes are often observed in the same types of habitats as prairie rattlesnakes during the summer months, possibly due to a shared reliance on small mammal prey (Kissner and Nicholson 2003b). However, bullsnakes and prairie rattlesnakes appear to show some resource partitioning on a macro-habitat scale, at least in southern Saskatchewan (Gardiner *et al.* 2013). Bullsnakes there show a preference for lowland pastures, hills/slopes and roads, whereas prairie rattlesnakes select for prairie dog colonies, hills/slopes and riparian areas. Both species avoided native upland habitat, cropland, open water and irrigated fields. On a micro-habitat scale, a best-fit model predicting bullsnake use of the landscape in southern Saskatchewan included the following variables: percent bare ground cover; percent shrub cover; maximum vegetation height; vegetation density; and proximity to burrows (Martino *et al*. 2012). Only distance to burrows was a significant positive predictor of habitat use by bullsnakes on its own.

In Wisconsin, a study of bullsnake habitat preferences showed that the species preferred open grassy slopes and avoided forests (both on slopes and on flat terrain) as well as agricultural areas (including cropland, tame pasture and ponds) (Kapfer *et al.* 2008b). Bullsnakes in this study also showed high habitat fidelity, returning to the same area year after year. *Pituophis catenifer deserticola* in Idaho showed no habitat preferences (Diller and Wallace 1996). However, in the south Okanagan Valley, they spend the active season in sandy and riparian habitats, using favourite retreat sites such as rodent burrows from which to base their foraging activities (Shewchuk 1997, cited in COSEWIC 2002).

3.3.5 Area Requirements

Bullsnakes in Saskatchewan appear to travel longer distances and occupy larger home ranges than other members of the species to the south in the United States (Martino *et al.* 2012). In southern Saskatchewan, the average home range size for bullsnakes was 87 ha (range: 38 ha to 151 ha), with males having a smaller home range than females (58 ha vs. 116 ha), although the sample size was too small to determine statistical significance (Martino *et al.* 2012). Bullsnakes in Saskatchewan travelled an average of 1.7 km from their hibernacula (range: 0.5 km to 3.9 km) (Martino *et al.* 2012). At C.F.B. Suffield, bullsnakes were reported to travel up to 12 km from their hibernacula along the South Saskatchewan River after emerging from hibernation (A. Didiuk, unpubl. data, cited in Kissner and Nicholson 2003b). Bullsnakes generally exhibit a dispersal pattern away from hibernacula along river slopes in May that is similar to that of prairie rattlesnakes (Didiuk 1999).

4 SHORT-HORNED LIZARD

4.1 Background

Short-horned lizards are greenish-brown coloured, dorso-ventrally flattened with a rim of protective spikes around the posterior margin of the skull and a fringe of protruding scales along their sides. The short-horned lizard, Alberta's only lizard species, is restricted to the extreme southeastern parts of the province (James *et al.* 1997). Short-horned lizards are associated with arid and semi-arid regions of west-central North America. Populations in Alberta are at the

northern periphery of their distribution in Alberta and occur in four disjunct locations in the province (James *et al.* 1997, James 2002, Environment Canada 2015). The total area comprises approximately 2,162 km^2, and includes: (1) the South Saskatchewan River valley (~639 km^2); (2) the Chin Coulee/Forty Mile Coulee complex (~231 km^2); (3) the Manyberries badlands area (~387 km^2); and (4) the valleys of the Milk River and Lost River (~905 km^2) (Powell and Russell 1998; SRD 2004; Environment Canada 2015). The short-horned lizard also occurs in four disjunct locations in and around Grasslands National Park in southern Saskatchewan (Environment Canada 2015). Short horned lizards also occur in the extreme south-central portion of Saskatchewan along the Frenchman and Poplar river watersheds (James *et al.* 1997).

Short-horned lizards are listed as 'At Risk' in Alberta under the provincial General Status listing (GoA 2016). The short-horned lizard is listed as 'Endangered' under the Alberta *Wildlife Act*. In 1992, COSEWIC designated the short-horned lizard as a species of 'Special Concern' in Canada (Powell and Russell 1992). The species was up-listed to 'Endangered' status in 2007 and added to Schedule 1 of the federal *Species at Risk Act* (*SARA*) (COSEWIC 2016). As a result, a *SARA* Recovery Strategy was developed in 2015 (Environment Canada 2015). The short-horned lizard is listed as a species of 'Least Concern' by the IUCN (Hammerson 2007).

Populations of short-horned lizards in Alberta are rare, low in abundance and localized (GoA 2016). Although more information is needed to assess population sizes and trends, population declines are suspected to have occurred in Alberta and the species is thought to occur at lower densities in the province relative to adjacent populations in the United States (James *et al.* 1997). While the number of short-horned lizard subpopulation appears to be decreasing, the subpopulations themselves appear to be relatively stable (GoA 2016).

Due to its low population density and cryptic habits, it is difficult to accurately assess populations of short-horned lizards. The most recent population inventories in Alberta were conducted in 2001 and 2002 (James 2002). These surveys encompassed all historically recorded short-horned lizard locations in the province. A total of 125 short-horned lizards were inventoried in 2001; the average density of lizards was calculated to be approximately 2 individuals/ha (James 2002). Based on short-horned lizard capture rates, populations in 2001-2002 generally appeared to be stable in comparison to previous search efforts (James 2002, James 2003). However, short-horned lizards were not observed at some previously occupied sites, such as the Comrey site within the Milk River Natural Area that was previously noted as being one of the most populous sites (Powell and Russell 1992, James 2002). These findings may indicate that local population extinction has occurred and that the distribution of this species in Alberta has contracted (James 2002); however, available data is currently insufficient to determine population trend (COSEWIC 2007). Once a short-horned lizard sub-population has been lost, it is very unlikely that recolonization will occur because of the limited dispersal abilities of the species and the great distances between known sites (James 2003). James (2003) estimated that there could be as few as 6,970 mature short-horned lizards in Alberta. However, due to lack of long-term population data, the estimate is considered very unreliable (James 2003). Surveys from Grasslands National Park from 2008 to 2011 resulted in a population estimate of 5,200 to 8,320 mature individuals for Saskatchewan (Fink, unpubl. data, cited in Environment Canada 2015). Ongoing population monitoring at select locations on an annual basis is required to provide long-term population trend information (James 2002).

Short-horned lizards are limited by both natural and human-related factors. As this species is at the northern edge of their range in Alberta, climatic constraints (*i.e.*, drought and severe winters) have a significant influence on their distribution and may affect productivity and survival rates. Predation is another natural limiting factor, however, losses due to predation are presumed to be minimal (James *et al.* 1997). Human-related threats include habitat loss due to cultivation, oil and gas exploration, roadway development and urbanization (James *et al.* 1997). Habitat degradation or disturbance by livestock grazing is considered a lesser threat (James *et al.* 1997).

4.2 Ecology

Short-horned lizards in Alberta generally emerge in mid- to late April and are active until mid-September when they begin to seek hibernacula, or overwintering sites (Powell and Russell 1996). Depending on weather conditions, they may remain moderately active at the overwintering site until late October or early November (Powell and Russell 1993).

Breeding occurs in mid to late May shortly after emergence from hibernation (Russell and Bauer 2000). Females produce one clutch of young per year (James *et al.* 1997). After mating, females remain at the coulee rims while male short-horned lizards are reported to disperse into the surrounding short-grass prairie (Russell and Bauer 2000). Females are viviparous and give birth to 6 to 11 young in late July to early August (Powell and Russell 1991, Russell and Bauer 2000). Gravid females are generally restricted to a small home range due to the high energy demand of carrying young (Russell and Bauer 2000). In Alberta, female short-horned lizards are thought to live for approximately five years; the lifespan of male lizards is unclear (James *et al.* 1997). Females breed in their second year, while males become sexually mature in their first year (James *et al.* 1997). There is a pronounced sexual dimorphism in Alberta short-horned lizard populations, with male lizards being approximately half the size of females (James 2002).

4.2.1 Diet

Short-horned lizards have a specialized diet consisting mostly of ants, but they may also eat a variety of other insects such as beetles and grasshoppers (Powell and Russell 1984). Ants are small and contain much indigestible chitin, so that large numbers of them must be consumed. Therefore, the short-horned lizard possesses a large stomach for its body size and a specialized digestive system (Pianka and Parker 1975). Typical prey sizes range from 2.1 mm to 6.0 mm (Powell and Russell 1984).

Short-horned lizards are small, well-camouflaged, sit-and-wait predators that remain motionless until prey comes within striking distance (Pianka 1966). Their main prey items, ants, are frequently clumped in distribution and so can be considered a patchy resource (Powell and Russell 1984). Competition for or shortage of this resource may force short-horned lizards to shift their diet to more of that of a dietary generalist (Powell and Russell 1984). It has been found that increases in the relative density of invertebrate prey species do not necessarily increase short-horned lizard populations (Reynolds 1979).

4.2.2 Predators

Short-horned lizard predators include loggerhead shrikes (*Lanius ludoviciana*), hawks, foxes, coyotes, striped skunks, and a variety of snakes (Smith 1993, Powell and Russell 1996). Short-horned lizards have flat, rounded bodies, reducing their speed and decreasing their ability to flee from predators (Pianka and Hodge 2002). As a result, natural selection has favoured camouflage, a spiny body form and cryptic behaviour rather than a sleek body and rapid movement as in the majority of other lizards. Risks of predation are likely to increase during long periods of exposure while foraging in the open.

4.3 Habitat Requirements

4.3.1 General

In Alberta, short-horned lizards occur only in the southeastern part of the province within the Dry Mixedgrass Subregion on sparsely-vegetated, south-facing slopes of coulees and canyons, at the interface between the prairie grassland and the coulee bottom (James *et al.* 1997). Powell and Russell (1993, 1998) described three different habitat types preferred by the short-horned lizard in Alberta: the Milk River Watershed, Bearpaw and North Marginal types. In the Milk River Watershed, preferred habitat tends to be at the ecotone between the native prairie upland and coulee and canyon margins, although short-horned lizards have reportedly been found in coulee bottoms and in open grassland areas as well. Short-horned lizards are also found along the southern perimeter of the Cypress Hills plateau on sandy dunes formed from Bearpaw shale, often matted with creeping juniper (*Juniperus horizontalis*) vegetation (the Bearpaw type). The North Marginal habitat type includes the northern-most populations of short-horned lizards in Alberta (around Medicine Hat), which occupy the south-facing rims of canyons and coulees.

Vegetation is an important aspect of short-horned lizard micro-habitat. Thinly-vegetated south-facing slopes appear to be preferred (Powell and Russel 1998). Short-horned lizards appear to be more influenced by habitat structure than vegetative species composition (James 2002). They also appear to require a relatively open soil surface (approximately 50% exposed soil), with some vegetative cover (James 2002). Shrub cover, primarily silver sagebrush and creeping juniper, may be more significant to adults, while grasses may be suitable for young short-horned lizards (James 2002). Selection of habitat is mostly dependent on suitable sites for thermo-regulation, foraging and overwintering.

The HSI model developed for short-horned lizards in the Milk River Watershed included four, equally-weighted variables: topographical features; native prairie class; elevation; and riparian zones (Taylor 2004). According to this model, optimal short-horned lizard habitat is restricted to uncultivated areas below 1,100 m that occur within 100 m of valleys (coulees), including valley breaks and bottoms but excluding riparian areas. Other variables that were considered for inclusion in the model were slope and aspect. Short-horned lizards appear to select moderately shallow to moderately steep slopes (10 to 60 degrees) at the microhabitat level. South-facing slopes are also typically selected, although lizards along the Milk River have been recorded on other aspects.

An RSF model was developed for short-horned lizard habitat preferences in southern Alberta for MULTISAR in 2010 (Skilnick *et al.* 2010). This modelling exercise tested numerous biophysical variables derived from such datasets as the Grassland Vegetation Inventory (GVI), Agricultural Region of Alberta Soil Inventory Database (AGRASID), quaternary geology and a digital elevation model (DEM) from which to derive slope, elevation and aspect. Available habitat was restricted to coulees and valleys in southeastern Alberta. Variables selected for the best-fit model included: slope aspect (northness and eastness); elevation; distance to Loamy ecological sites; distance to Badlands ecological sites; distance to native upland sites; distance to riparian areas; distance to fluvial deposits; percent grass cover; percent non-vegetation cover; and percent shrub cover. Within the confines of coulees and valleys, short-horned lizards appear to be selecting for south-east-facing slopes; higher elevations; proximity to Loamy and Badlands ecological sites and fluvial deposits; greater distances from riparian areas and native upland sites; and habitats with a mix of grass, shrubs and bare ground/bedrock.

4.3.2 Overwintering Habitat

It was initially believed that short-horned lizards utilized crevasses within rock underlying Bearpaw shale to overwinter. However, radio telemetry has shown that short-horned lizards excavate small burrows, approximately 10 cm beneath the surface, in loose soil of south-facing slopes and rely on snow cover for insulation (James *et al.* 1997). In Colorado, short-horned lizards do not necessarily select for south-facing slopes in which to overwinter; instead, they selected for steep wash banks with suitable (bare and penetrable) soil within their home range (Mathies and Martin 2008). Soil for over-wintering needs to be thin and friable to enable excavation by short-horned lizards (SRD 2004). Soil around the base of shrubs that has been loosened by the rooting action of the plant may provide suitable burrowing and overwintering sites. Low short-horned lizard population densities in Alberta may be the result of high overwintering mortality as a result of seasonal extremes (James *et al.* 1997, Powell and Russell 1996).

4.3.3 Thermoregulation

Like prairie rattlesnakes and bullsnakes, short-horned lizards are typically found on moderately steep, south-facing slopes (James 2002). These slopes are utilized for their maximum exposure to solar radiation needed to help regulate body temperature. Like other reptiles, short-horned lizards use a behaviour known as “shuttling” where they move between heat sources and heat sinks to control their internal body temperature within a preferred range (Powell and Russell 1985, James *et al.* 1997). By positioning their body in a certain orientation while basking in direct sunlight they are able to increase the surface area to incident radiation. “Thigmothermy”, or the process of gaining heat from an object (such as a rock), has been observed in short-horned lizards (James *et al.* 1997). Habitat use is generally more varied on sunny days (Powell and Russell 1985). Shelter, provided by shade from shrubs or mammal burrows, is required to cool body temperature. Individuals may also burrow into the substrate to seek relief from the heat (Heath 1964). A loose substrate which short-horned lizards can burrow into to seek shade or retreat at night is therefore important (Powell and Russell 1996). Vegetation cover is also used to provide overnight cover (James *et al.* 1997).

4.3.4 Foraging Habitat

Short-horned lizards forage on thinly vegetated south-facing slopes, although lizards have been found on east-, west- and some north-facing slopes as well (James *et al.* 1997). Thick vegetation in riparian habitats generally impedes travel through these areas due to the low profile and short legs of short-horned lizards (James *et al.* 1997). However, risk of predation also tends to increase during long periods of exposure while foraging in the open.

4.3.5 Area Requirements

During the summer, short-horned lizard daily movement patterns rarely exceed 30 m and mostly occur along the slopes of the valleys or valley bottom (Powell and Russell 1996). Radio-telemetry data indicated that short-horned lizards in Alberta do make forays of approximately 70 m from the valley break into the adjoining prairie (Powell and Russell 1996). Female short-horned lizards generally travel furthest prior to mating (up to 100 m or more), after giving birth (usually less than 50 m) and prior to hibernation (up to 266 m) (James *et al.* 1997; James unpubl. data, cited in SRD 2004).

Reported home ranges for short-horned lizards in Alberta vary from 4 m^2 to 2,400 m^2 (James *et al.* 1997) and may possibly be as high as 4,000 m^2 (Powell and Russell 1996). Females are known to return to similar home range sites annually (Powell and Russell 1994). Males appear to have much larger home ranges than females and disperse over long distances during the active season (Powell and Russell 1996). Although short-horned lizards are not territorial, studies in Alberta have found little evidence of home range overlap (Powell and Russell 1996).

5 YELLOW-BELLIED RACER

5.1 Background

The eastern yellow-bellied racer, a type of snake, is Alberta's only known species of racer. It has a long, slender body that is blueish, blueish-green or grey with a bright yellow belly. The racer is the only species of *Coluber* in North America, where it has been divided into 11 subspecies (Wilson 1978). Three of the subspecies occur in Canada: *Coluber constrictor foxii* (blue racer), which occurs only in Ontario; *C. c. mormon* (western yellow-bellied racer), which occurs only in British Columbia; and *C. c. constrictor* (eastern yellow-bellied racer), which occurs in Saskatchewan and Alberta. Racers are non-venomous and do not pose a risk to humans, although they may bite if cornered.

The eastern yellow-bellied racer is at the northern-most limit of its range in southern Canada. As a result, population sizes are small and isolated due to a lack of suitable habitat (COSEWIC 2004). The eastern yellow-bellied racer was previously thought to occur only in Saskatchewan, where it occurs in the extreme southern part of the province in the Big Muddy and Frenchman River valleys and possibly in the Cypress Hills (COSEWIC 2004, Gardiner *et al.* 2011). There are only seven known recorded locations of racers in Saskatchewan (COSEWIC 2004). Based on the restricted distribution and limited number of captures, the population in Saskatchewan is estimated at fewer than 10,000 individuals (COSEWIC 2004). There have only been a handful of

eastern yellow-bellied racer sightings in Alberta, all in the far southeastern part of the province in the Onefour and Lost River areas (Parks Canada Agency 2010, Gardiner *et al.* 2011, MULTISAR 2014). There have been no empirical studies of racer population numbers in Alberta.

The racer is listed as 'Accidental/Vagrant' under the Alberta General Status listing (GoA 2016). The eastern yellow-bellied racer was originally listed as a species of 'Special Concern' in Canada in 1991 (Campbell and Perrin 1991) and then upgraded to 'Threatened' in 2004 (COSEWIC 2004). The subspecies is currently listed as 'Threatened' by COSEWIC (2016) and included as a 'Threatened' species under Schedule 1 of *SARA*. A *SARA* Recovery Strategy was developed for the eastern yellow-bellied racer in 2010 (Parks Canada Agency 2010). *Coluber constrictor* is listed as a species of 'Least Concern' by the IUCN (Hammerson *et al.* 2013).

Because of its small size and restricted distribution, eastern yellow-bellied racers are extremely vulnerable to extirpation and genetic isolation due to large mortality events such as major slumping events, as occurred at one site in southern Saskatchewan in 2011 (Gardiner and Sonmor 2011).

5.2 Ecology

Hibernacula are required by eastern yellow-bellied racers to overwinter in Canada. Emergence from hibernacula is typically in April and May in Canada, with racers returning to their hibernacula in September and October prior to the onset of winter temperatures (Parks Canada Agency 2010). Courtship and reproduction data are limited for eastern yellow-bellied racers in Canada; however, in Kansas they mate in spring, and it is likely that individuals from Saskatchewan mate after emerging from their dens in spring as well (Fitch 1963; COSEWIC 2004). Mating in *Coluber constrictor mormon* occurs in May away from their winter dens on their summer breeding grounds (Shewchuk and Waye 1995).

Male racers generally reach sexual maturity at 11 months of age, compared to two to three years of age for females (Parks Canada Agency 2010). Racers are promiscuous, with two or more males often simultaneously courting the same female (Fitch 1963). Eastern yellow-bellied racers generally have an even sex ratio of 1:1, although this may change with age with fewer males surviving to advanced age than females (Fitch 1963, Maccartney pers. comm., cited in Campbell and Perrin 1991). Clutch sizes for *Coluber constrictor* range from three to 20 eggs, with larger clutch sizes common near the northern limits of its range (Fitch 1963; Maccartney pers. comm., cited in Campbell and Perrin 1991; Rosen 1991; COSEWIC 2004). Incubation in *C. c. mormon* lasts from 40 days to two months, with length of time depending on temperature (Nussbaum *et al.* 1983). *C. c. mormon* females lay their eggs in June or July, and hatchlings will begin to emerge in late July to August (Nussbaum *et al.* 1983). Following reproduction, females and males disperse throughout their summer habitat.

5.2.1 Diet

In Canada, eastern yellow-bellied racers are opportunistic hunters with a broad prey base (Parks Canada Agency 2010). Common prey items include crickets, grasshoppers, other insects, spiders,

small rodents, lizards (and their eggs), amphibians and, in some instances, juvenile snakes (Parks Canada Agency 2010). Racers in other parts of its range prefer voles (*Microtus* spp.), white-footed mice (*Peromyscus* spp.), lizards, snakes, frogs, toads and soft-bodied insects (*e.g.*, grasshoppers) (Fitch 1963). In British Columbia, one study found that more than 90% of the diet of *Coluber constrictor mormon* individuals was comprised of insects from the family Orthoptera (Grasshoppers) (Shewchuk and Austin 2001).

5.2.2 Predators

Fitch (1963) estimated that a healthy breeding population of 100 adults with a 1:1 sex ratio could produce about 300 eggs, of which only 50 (17%) would survive past the first year of life. Racers live to approximately seven or eight years old in the wild (Fitch 1963). In other parts of its range, crows (*Corvus* spp.), marsh hawks (*Circus cyaneous*), kestrels (*Falco* spp.) and broad-winged hawks (*Buteo platypterus*) prey on *Coluber constrictor* (Fitch 1963). In Saskatchewan, both a Swainson's hawk (*Buteo swainsoni*) as well as a coyote (*Canis latrans*) or an American badger (*Taxidea taxus*) were found to have preyed on racers (Maccartney pers. comm., cited in Campbell and Perrin 1991). In British Columbia, available circumstantial evidence suggests that predators of *C. c. mormon* include falcons (*Falco* spp.), hawks (*Buteo* spp.), American badgers and striped skunks (*Mephitis mephitis*) (Maccartney pers. comm., cited in Campbell and Perrin 1991).

5.3 Habitat Requirements

5.3.1 General

Racers generally prefer open habitat, including prairies, meadows, hillsides, and pastures (Fitch 1963). In Canada, racers prefer mixedgrass prairie and sagebrush thickets (COSEWIC 2004; Parks Canada Agency 2010). *Coluber constrictor flaviventris* will also inhabit forested areas in parts of its range in the United States (Fitch 1963). *C. c. mormon* also prefers open grassland habitat, but it can be found in open-canopied forests as well (COSEWIC 2004). On a landscape scale, eastern yellow-bellied racers are typically found along river valleys in Saskatchewan (Martino *et al.* 2012). In the Osoyoos region of British Columbia, western yellow-bellied racers are often found in wet valley bottoms as well as valley sides, including sandy terraces and rocky slopes (Shewchuk and Waye 1995).

5.3.2 Hibernacula

Racers typically show fidelity to their overwintering dens year after year (COSEWIC 2004). However, it has been reported that the snakes will occasionally switch hibernacula for no known reason (Fitch 1963). In Saskatchewan, eastern yellow-bellied racers have been found over-wintering in communal dens with prairie rattlesnakes, bullsnakes, plains garter snakes, and rarely, plains hognose snakes (Kissner *et al.* 1996, Wright and Didiuk 1998, Gardiner *et al.* 2011, Gardiner *et al.* 2013).

Racer hibernacula need to be suitably deep underground to prevent freezing over winter. Suitable denning sites include: deep holes in soft soil on hillside slopes; mammal burrows; deep crevices in or near limestone rock ledges; hillside slumps; and abandoned cisterns (Fitch 1963; Maccartney pers. comm., cited in Campbell and Perrin 1991, COSEWIC 2004, Parks Canada

Agency 2010). It was originally thought that southerly aspects were preferred hibernacula sites (Maccartney pes. comm., cited in Campbell and Perrin 1991); however, recent evidence suggests that a greater variation in aspects may be utilized than originally thought, although north aspects are generally avoided (Parks Canada Agency 2010).

Eastern yellow-bellied racer hibernacula characteristics have not been described in the literature. However, hibernacula have been described for western yellow-bellied racers in British Columbia. Given the similarities between the two subspecies, hibernacula needs may be expected to be similar. Hobbs and Sarrell (2002) described four main attributes that suitable western yellow-bellied racer overwintering sites possessed:

- Fracturing – Suitable denning sites have rock fracturing, which allows geothermal heat to rise up and maintain a temperature in the den between 4°C and 9°C.
- Humidity – Because racers are prone to desiccation, sufficient humidity may be required to prevent snakes from becoming dessicated.
- Cover – Overwintering sites often have cover in the form of boulders, talus, and brush, which aids in thermoregulation during emergence.
- Thermal momentum – Hibernacula require a certain capacity to absorb and retain heat. This capacity is influenced by slope, aspect, mass, position, and surface albedo.

5.3.3 Summer Habitat

Grassland communities with short vegetation structure, such as over-grazed pastures, mowed and burned areas, lack food and shelter for racers and, therefore, do not provide adequate habitat on their own (Fitch 1963). Such habitats may be acceptable provided there are patches of brush or other dense vegetation nearby. Retreat sites such as burrows and shrubs are considered critical for racer survival (Martino *et al*. 2012). Cropland and hay fields provide suitable habitat for racers when vegetation is tall, but when vegetation is short (*e.g.*, early in the year or after being cut), racer numbers will depend on the availability of refugia in nearby pastures, thickets and/or forests (Fitch 1963). Eastern yellow-bellied racers in Kansas rely on a mix of grassy areas and shrubby patches within native tallgrass prairie (Klug *et al.* 2011).

In their Saskatchewan study, Martino *et al.* (2012) found that, on a macro-habitat level, eastern yellow-bellied racers used lowland pastures, mudflats and riparian areas 2 to 5 times more than would be expected based on their availability in the landscape. Roads and hills/slopes were used as they would be expected based on their availability. Native upland sites were used 12 times less and irrigated hay fields were used 2 times less than expected. Racers were not observed in cropland, open water or prairie dog colonies in the study. Subsequent research published by the same authors supported the preference for riparian areas, lowland pastures and mudflats (Gardiner *et al*. 2013). In addition, this subsequent research suggested that hills/slopes were also selected for, and that together with riparian areas, were used more than lowland pastures and mudflats (Gardiner *et al*. 2013). Prairie dog colonies were strongly avoided by racers in this second study.

On a micro-habitat level, Martino *et al.* (2012) found that a model containing the variables percent shrub cover, vegetation density, proximity to burrows and proximity to shrubs was the

best predictor of racer use on the landscape. Racers selected for higher shrub cover, higher vegetation density, and greater proximity to burrows; proximity to shrub cover was not a significant predictor on its own.

5.3.4 Egg-laying Habitat

Eggs are laid in loose soil, mammal burrows, or under large rocks and left to incubate under ambient heat (Parks Canada Agency 2010).

5.3.5 Area Requirements

In Kansas, Fitch (1963) estimated population density of *Coluber constrictor flaviventris* at 0.45 individuals/ha to 1.1 individuals/ha, with better habitat supporting as many as 1.2 individuals/ha. Klug *et al.* (2011) estimated the home range sizes of *C. c. flaviventris* individuals in the Flint Hills of Kansas and Oklahoma to be approximately 11.5 ha (range: 1.2 ha to 33.5 ha). Male home range sizes were 3.5 times larger than for females (average = 21.6 ha vs. 6.4 ha) (Klug *et al.* 2011). Population density of *C. c. mormon* was variously estimated at 0.65 individuals/ha and 0.32 individuals/ha in Utah (Brown 1973 and Brown and Parker 1984, both cited in COSEWIC 2004). Population of *C. c. mormon* in British Columbia was estimated to be lower than this, except in the South Okanogan area where there is better habitat (Shewchuk and Waye 1995).

In Saskatchewan, eastern yellow-bellied racer home ranges averaged 145 ha, with a range of 11.2 ha to 714.4 ha (Martino *et al.* 2012). Home range sizes varied considerably by sex: males had an average home range size of 192 ha (n=8) and females had an average home range size of 83 ha (n=6). Racers in Saskatchewan dispersed an average of 2.5 km from their den sites (range: 0.8 km to 5.0 km) (Martino *et al.* 2012). These results suggest that racers in Saskatchewan have much larger home range sizes and disperse much greater distances from their hibernacula than their southern cousins (Martino *et al.* 2012). Home ranges tend to be ‘dumbbell-shaped’, with two activity areas centred on the hibernacula and summer range connected by a narrow movement corridor (Gardiner *et al.* 2013).

6 GRAZING AND HERPETILE GROUP 1

Livestock grazing is the most common human land-use over much of the area in which prairie rattlesnakes, bullsnakes, short-horned lizards and eastern yellow-bellied racers are found in Alberta (James *et al.* 1997, Watson and Russell 1997, Kissner and Nicholson 2003b, James pers. comm.). Livestock grazing has the potential to affect these reptiles directly either due to disturbance or trampling of adults, young or eggs; through direct physical damage to critical habitats; or due to removal of important shelter, thermal or escape vegetation cover (Romero-Schmidt *et al.* 1994). Indirectly, livestock herbivory affects plant species composition and vegetation structure, which may influence the community composition or abundance of small mammal or insect prey. Few studies have been done to examine the influence of grazing on these reptiles in Alberta, or elsewhere in their range. The following discussion summarizes the potential interactions of grazing on prairie rattlesnakes, bullsnakes, short-horned lizards and eastern yellow-bellied racers based on available studies and discussions with species experts.

6.1 Prairie Rattlesnake and Bullsnake Response to Grazing

Livestock grazing is generally not thought to heavily impact prairie rattlesnake or bullsnake hibernacula as these sites are typically located away from riparian corridors, on moderate to steep, well-drained, xeric, south-facing slopes with low forage value (Kissner pers. comm., Nicholson pers. comm.). Hills and slopes are also preferred summer habitat for bullsnakes and prairie rattlesnakes, among other habitat types (Gardiner *et al.* 2013). Cattle tend to avoid rough terrain and steep slopes and typically remain in flatter upland or lowland areas, most often concentrating use near riparian areas (Van Vuren 1982, Fitch and Adams 1998). However, in order to access riparian areas in valley bottoms, cattle create access trails across steep slopes and will repeatedly use the same trails. If these access trails concentrate trampling and grazing pressure near hibernacula, slope erosion may result that has the potential to cover up or compress soils at the openings of snake hibernacula (Didiuk 1999). Heavy use along south-facing valley slopes can also impede openings of bullsnake nesting burrows if trampling occurs near burrow openings (Didiuk 1999). Grazing along slopes near hibernacula also has the potential to reduce vegetative cover used by gravid female prairie rattlesnakes and their young (Didiuk 1999). Grazing practices that conserve shrub cover (particularly silver sagebrush) and limit cattle access near hibernacula will benefit prairie rattlesnakes, bullsnakes and racers. Ensuring sufficient shrub cover is retained near hibernacula is important for improving the availability of shade, thermal and predator escape cover for snakes (Didiuk 1999). As snakes are found near dens in the spring and fall, avoiding grazing near hibernacula at these times is considered especially important (Nicholson pers. comm.).

The most significant impact of grazing on prairie rattlesnakes and bullsnakes may be associated with prey abundance in upland native prairie within 15 km of hibernacula (Kissner pers. comm., Nicholson pers. comm.). Grazing practices that promote a greater diversity and abundance of small mammals and retain sufficient cover in foraging sites are considered advantageous to both snake species (Nicholson pers. comm.). Consistent heavy grazing over time has been found to reduce small mammal species diversity and depress populations (Rosenzweig and Winakur 1969, Fagerstone and Ramey 1996). A reduced prey base forces snakes to travel further from hibernacula to forage, making them more susceptible to predation or road mortality (Nicholson pers. comm.). If snakes are unable to accumulate sufficient reserves over the active season they may also compromise their winter survivability (Watson and Russell 1997). Meadow voles and northern pocket gophers, two important prairie rattlesnake and bullsnake prey items, have both been correlated with increasing plant biomass and generally occur in higher densities in light to moderately grazed prairie (Fagerstone and Ramey 1996, Reynolds *et al.* 1999). Other important prey species, such as Richardson's ground squirrels, occur in higher densities in more heavily grazed areas (Fagerstone and Ramey 1996, Michener and Schmutz 2002). Grazing practices that promote vegetation heterogeneity are likely most beneficial to these two snake species as this will promote a more stable and varied prey base.

Riparian area grazing may need to be controlled in prairie rattlesnake and bullsnake home ranges. Although prairie rattlesnakes may forage primarily in upland native prairie (Gannon and Secoy 1985, Nicholson pers. comm.), recent research suggests that the species also makes extensive use of riparian areas during the summer, at least in Saskatchewan (Gardiner *et al.* 2013). Therefore, summer grazing along riparian areas has the potential to negatively impact

prairie rattlesnakes. As bullsnakes are known to commonly use treed areas along river valleys for foraging, they may also be susceptible to riparian grazing effects (Kissner and Nicholson 2003b).

6.2 Short-horned Lizard Response to Grazing

The impact of livestock grazing on short-horned lizard habitat and populations in Alberta is believed to be relatively minimal and may be comparable to the impact bison may have had in the past (Powell and Russell 1993). In general, as short-horned lizards generally inhabit areas of rough terrain, such as steep hillsides and juniper dunes, they tend to occur in areas that receive little use from cattle (James pers. comm.). According to Powell and Russell (1993), impacts from cattle grazing that are most likely to negatively affect short-horned lizards are physical damage near coulee edges as well as seeding of invasive species such as crested wheatgrass (*Agropyron pectiniforme*) that may restrict their movements.

As with prairie rattlesnakes and bullsnakes, the possible impacts of livestock use on short-horned lizards may be greatest along heavily used access trails into valley bottoms that traverse south-facing slopes. Trampling and soil compression in overwintering areas may make it more difficult for lizards to burrow into the ground. Due to their tendency to remain motionless while foraging or as a predator avoidance strategy, short-horned lizards are more likely to be vulnerable to direct trampling than are snakes (James pers. comm.). Short-horned lizards may be particularly vulnerable to trampling during late July to early August as this is when young lizards are born, and lizard densities are at a peak (James pers. comm.). Due to severe drought conditions in 2001 to 2002, stocking rates were either severely reduced or cattle were not grazed during the short-horned lizard survey that was conducted in Alberta at this time (James 2002, James pers. comm.). Therefore, the effect of cattle use in or near short-horned lizard habitats could not be assessed at that time (James pers. comm.).

Various studies in southern Texas (Burrow *et al.* 2001), southeastern Idaho (Reynolds 1979), western Nebraska (Ballinger and Jones 1985), western Arizona (Jones 1981) and Baja California (Romero-Schmidt and Ortega-Rubio 1999) have examined the influence of livestock grazing on other horned lizard subspecies or other lizard species. Burrow *et al.* (2001) found that habitat selection of Texas horned lizards (*Phrynosoma cornutum*) in native prairie of southern Texas did not differ with land management treatments (*i.e.*, burning and grazing). Burrow *et al.* (2001) noted that management practices that maximize the availability of a suitable mosaic of bare ground, herbaceous vegetation and woody vegetation in close proximity may lead to higher lizard densities. Reynolds (1979) found that Pygmy short-horned lizards (*Phrynosoma douglassi*) responded positively to the reduced vegetative cover caused by sheep grazing due to an increase in availability of basking sites. Pygmy short-horned lizards were most abundant in sheep-grazed areas dominated by big sagebrush (*Artemisia tridentata*). Ballinger and Jones (1985) also reported a positive relationship between grazing and the maintenance of habitat for two lizard species that occur primarily in open Blowout habitats in the sandhill prairies of western Nebraska. Reynolds (1979) found that conversion of native habitat to crested wheatgrass tame pasture resulted in a significant decrease of pygmy short-horned lizards.

Jones (1981) found that grazing exerted a negative impact on lizard abundance and diversity in western Arizona only when it was associated with significant changes in vegetation structure.

Romero-Schmidt and Ortega-Rubio (1999) found that three lizard species exhibited differing responses to grazing in a desert scrub habitat. The 'sit-and-wait' type predator lizard was most negatively affected by grazing due to losses in perennial grass cover and a corresponding reduction in the abundance of certain invertebrates (Romero-Schmidt and Ortega-Rubio 1999). As discussed previously, short-horned lizards have a specialized diet consisting mostly of ants. Intensive grazing by livestock has been shown to have adverse effects on seed-harvester ant species such as *Pogonomyrmex desertorum* due to removal of flowering tillers on grasses and trampling effects on herbaceous annuals. Shrub encroachment, however, can result in increased relative abundance of liquid-feeding ant species (Nash *et al.* 2000).

6.3 Eastern Yellow-bellied Racer Response to Grazing

Eastern yellow-bellied racer hibernacula often occur on steep slopes, and the species shows a preference for hills and slopes (Gardiner *et al.* 2013). Livestock tend to avoid slopes unless travelling through the area. As discussed above with respect to bullsnakes and prairie rattlesnakes, trampling and erosion are potential concerns to snake hibernacula if livestock make use of slopes while travelling between preferred habitats. Such activity may result in filling in of hibernacula entrances (Parks Canada Agency 2010). Similar to prairie rattlesnakes and bullsnakes, avoidance of early season grazing near hibernacula sites is likely to benefit racers.

Gardiner *et al.* (2013) showed that summer habitat away from hibernacula also needs to be considered when managing racers. Heavy livestock grazing could result in loss of summer habitat for the eastern yellow-bellied racer (Maccartney pers. comm., cited in Campbell and Perrin 1994); however, negative effects of livestock grazing on racers have yet to be demonstrated (COSEWIC 2004). Because racers show a preference for riparian areas, mudflats and lowland pastures (Martino *et al.* 2012, Gardiner *et al.* 2013), heavy grazing in these areas (as well as on slopes) is likely to be detrimental to the species. Riparian areas are especially prone to over-use by livestock if not properly managed due to abundance of forage, water and shelter (Ohmart 1996). Therefore, proper riparian area management (*e.g.*, through corridor fencing or off-stream water development) may be crucial for racer survival. Because of its preference for high shrub cover and high vegetation density (Martino *et al.* 2012), grazing practices that promote these vegetation characteristics in native prairie are likely to benefit *Coluber constrictor flaviventris*. Light to moderate grazing in mixedgrass prairie promotes high vegetation cover (Adams *et al.* 2013a). Land management practices that promote high numbers of ground squirrel (*Spermophilus* spp.) and/or American badger burrows will likely increase the value of the habitat to racers (Martino *et al.* 2012) and other species (see raptor discussion).

7 GRAZING SYSTEMS AND HERPETILE GROUP 1

Table III-13 provides an overview of seven grazing systems and their potential implications for maintaining or enhancing habitat for prairie rattlesnakes, bullsnakes, short-horned lizards and eastern-yellow-bellied racers. A grazing system is a tool used to control the timing, intensity and frequency of livestock grazing (Holechek *et al.* 2003).

Table III-13 Grazing Systems and Herpetile Group 1 Habitat Management

Grazing System	Discussion
Continuous (Season-Long) Grazing	
Advantages:	Under light to moderate stocking rates, continuous grazing often leads to patch grazing effects in native prairie, with areas that are consistently re-used and other areas that receive little use. Variably grazed patches may help to create a heterogeneous habitat mosaic. The size and number of patches tends to increase as the season progresses, likely in response to increased grazing pressure (Ring *et al.* 1985). Grazed patches provide short grass habitat that is preferred by Richardson's ground squirrel's (Michener and Schmutz 2002), an important prairie rattlesnake and bullsnake prey. Once established, ground squirrel colonies may be capable of maintaining short grass habitat. Prairie rattlesnakes and bullsnakes are known to remain in areas and exploit patches with an abundance of colonial burrowing prey species (Hill *et al.* 2001). Grazed patches with reduced litter and reduced grass density may also provide unimpeded movement for migrating snakes, depending on patch size. Sagebrush and cacti, both considered unpalatable to cattle, may increase under continuous moderate grazing, increasing the availability of shade and predator escape cover for snakes. Unused patches of vegetation with greater litter cover or more vigorous plant biomass will provide habitat for other snake prey such as meadow voles and northern pocket gophers. A combination of variably grazed patches and unused areas thereby offers habitat to a greater diversity of small mammal prey, likely increasing the stability of this prey base. A heterogeneous physical environment may also provide more habitat patches within which short-horned lizards can forage (Powell and Russell 1984). Habitat management that creates a mosaic of bare ground, herbaceous vegetation and woody vegetation in close proximity will likely benefit short-horned lizards (Burrow *et al.* 2001). By reducing forb and grass cover, grazing in sagebrush habitats increases the availability of basking sites for short-horned lizards (Reynolds 1979, James 2002). Short-horned lizards in Alberta tend to use patches with at least 50% exposed soil (James 2002). An increase in sagebrush or shrub cover as a result of grazing may also be beneficial to short-horned lizards that rely on this cover for shade and overnight thermal cover. Loosened soil around the base of shrubs is thought to provide suitable sites for burrowing into for overwintering (James *et al.* 1997). Availability of sagebrush or shrub cover may also benefit populations of liquid-feeding ant prey (Nash *et al.* 2000).

Disadvantages:	Riparian areas are often heavily impacted by persistent cattle use under continuous grazing systems, particularly if alternate water supplies or fencing are not in place. Continuous grazing nearly always results in overuse of portions of riparian areas (Ohmart 1996, Fitch and Adams 1998). Overuse of riparian areas has the potential to impact stability of banks and slopes associated with waterways. Over-use of riparian and lowland areas adjacent to riparian habitat has the potential to directly impact snake foraging habitat (Martino *et al.* 2012, Gardiner *et al.* 2013). Slumping of banks and slopes may impact snake hibernacula or bullsnake nesting sites depending on the position of these sites on the slope (Kissner pers. comm.). However, little information is available about bullsnake nesting sites, and only a few sites have been found in Alberta (Kissner pers. comm.). Grazing during the spring and fall when snakes are basking near hibernacula may increase the vulnerability of snakes to predators if it significantly reduces important cover habitat around the hibernacula (Nicholson pers. comm.). This is particularly a concern in grass terrace habitats along river floodplains and escarpments (Didiuk 1999). Didiuk (1999) reported that prairie rattlesnakes and bullsnakes at C.F.B. Suffield remained near hibernacula for prolonged periods in the spring and fall. Under continuous grazing, cattle are often grazed throughout the growing season beginning in the spring and continuing throughout the summer and fall periods. Reductions in silver sagebrush density and canopy cover may occur under heavy stocking rates with confined livestock feeding (Adams *et al.* 2004).
Rotational Grazing– Deferred Rotation	
Advantages:	Deferred spring grazing has the advantage of minimizing possible grazing or trampling disturbance to river banks and slopes when high water tables and runoff make them more susceptible to erosion. This minimizes erosion and slumping risks to hibernacula and nesting sites. Grazing on mixedgrass prairie that defers use until after seed set would likely promote habitat use by small granivorous mammals that prairie rattlesnakes and bullsnakes prey upon (Fagerstone and Ramey 1996).
Disadvantages:	If grazing were deferred long enough in the growing season, the increased height and density of vegetation may discourage use by burrowing mammals, particularly Richardson's ground squirrels, and impede snake movement through the prairie.

Complementary Grazing	
Advantages:	As with deferred grazing systems, complementary grazing allows for improved plant vigour in grasses and increases the likelihood of seed set. Using tame pasture in the spring to defer the use of native prairie where river systems occur will allow banks to stabilize following spring runoff before cattle graze these areas.
Disadvantages:	Complementary grazing requires the use of tame pasture, which may not be available in all grazing systems. Tame pasture is often not suitable in many places in the Mixedgrass and Dry Mixedgrass Natural Subregions due to fragility of soils and dry conditions. Seeding native prairie is also not desirable as the taller growth form of most tame forage species may impede snake and lizard movements. Reynolds (1979) noted a significant decrease in abundance of short-horned lizards in sagebrush prairie that had been converted to crested-wheatgrass in southeast Idaho. If implementation of a complementary grazing system requires the conversion of native prairie to tame pasture it is not likely to provide added benefit to Alberta herpetiles and will result in the loss of critical native prairie habitat.
Rotational Grazing	
Advantages:	Rotational grazing systems including switchback, deferred-rotation and rest rotation grazing allow for timed sequences of grazing and rest periods in smaller sized pastures (Holechek *et al.* 2003). Rotational grazing reduces selective grazing by encouraging use of plant groups differing in their season of growth and allows for seed production, seedling establishment and restored plant vigour (Adams *et al.* 1991). Vigorous grasses with high seed production attract small granivorous mammals such as mice (Fagerstone and Ramey 1996). Richard's ground squirrels consume leaves and stems of grasses, and a variety of forbs in early spring and summer, but seeds are their principal foods in late summer and autumn (Quanstrom 1968). Vegetation on south facing slopes of river valleys with moderate to steep gradients, the preferred habitat of short-horned lizards, is generally maintained by topography. Rapid drainage and erosion keep vegetation fairly sparse. However, excessive use by livestock may destabilize these slopes. Periodic rest from grazing with rotational grazing systems allows vegetation to maintain vigour and deep roots to prevent the acceleration of erosion. Rotational grazing systems also offer a means to control the timing of grazing in critical habitats to avoid sensitive periods. This includes avoiding use near hibernacula in the spring and fall and avoiding use near short-horned lizard birthing areas in late July to early August.

Disadvantages:	A noted disadvantage due to rotational grazing is the creation of uniform grazing effects due to improved cattle distribution. Uniform grazing effects are accentuated when high stocking densities are used, forcing cattle to use the entire area available for grazing. Prairie rattlesnakes tend to favour heterogeneous grassland conditions with high species diversity (Gannon 1978). Vegetation heterogeneity increases small mammal prey species diversity for snakes and is also important for promoting a diverse invertebrate prey base for short-horned lizards (Fagerstone and Ramey 1996, Jonas *et al.* 2002).
Rotational Grazing – High Intensity Grazing	
Advantages:	Intensive grazing in upland foraging sites may reduce the cover of vegetation, increasing the occurrence of some snake prey species such as Richardson's ground squirrels. Heavy grazing will also increase the relative composition of some shrub species, which are used by snakes for retreat sites.
Disadvantages:	Intensive grazing at high stocking rates tends to reduce biodiversity of the area, thereby reducing the variety of potential prey for snakes (Bai *et al.* 2001). In particular, consistent heavy grazing is known to depress populations of most small mammals, especially species such as meadow voles and northern pocket gophers that require greater vegetation cover (Fagerstone and Ramey 1996). The potential for negative impacts to riparian areas is also heightened with high stocking rates. Heavy use in and around riparian areas increases the probability of slumping, erosion and soil compaction that may occur in the vicinity of hibernacula or bullsnake nesting sites (Didiuk 1999). Overwintering sites are often located on slump zones and are therefore prone to erosion (Nicholson and Rose 2001). Heavy use of riparian areas also negatively impacts snake foraging habitat (Martino *et al.* 2012, Gardiner *et al.* 2013). Heavy use also increases trampling risks to short-horned lizards (James pers. comm.). Silver sagebrush also tends to decrease in cover under prolonged heavy grazing (Adams *et al.* 2004).
Riparian Pasture Grazing	
Advantages:	Hibernacula are often associated with sagebrush patches along river valleys (Nicholson and Rose 2001). The silver sagebrush/western wheatgrass (*Pascopyrum smithii*) habitat type in riparian areas is a disturbance-caused (disclimax) habitat type where site potential has been altered by prolonged heavy grazing (Thompson and Hansen 2002). Therefore, continued grazing in valley bottoms next to riparian areas may be necessary to maintain this habitat type.

	Riparian areas are important foraging areas for bullsnakes and eastern yellow-bellied racers and are also used sporadically by prairie rattlesnakes (Didiuk 1999; Martino *et al.* 2012). Well-managed riparian areas may result in sustained production of woody vegetation, such as cottonwoods, that are used by bullsnakes as foraging habitats. Avoiding grazing in the spring when banks are vulnerable to erosion may benefit the conservation of suitable hibernacula. Trailing action and grazing by cattle, especially in spring, tends to destabilize slopes. Short-horned lizards generally do not use riparian areas as thick vegetation impedes their movements (Taylor 2004, James pers. comm.).
Disadvantages:	Restricting use of riparian areas may concentrate heavier grazing in upland habitats and lead to uniform grazing effects or diminished habitat opportunities for a diverse small mammal prey base for snakes.

8 BENEFICIAL MANAGEMENT PRACTICE RECOMMENDATIONS

Grasslands, sand dunes, shrublands and river escarpments in the Milk River and South Saskatchewan River watersheds provide important and unique habitat for prairie rattlesnakes, bullsnakes, short-horned lizards and eastern yellow-bellied racers (Didiuk 1999). Maintaining critical native prairie habitats, in particular, minimizing disturbance to south-, southeast- and east-facing slopes that provide overwintering habitat, is important to the long-term conservation of these reptile species.

The following general land use and grazing recommendations offer a variety of means to protect or maintain prairie rattlesnake, bullsnake, short-horned lizard and eastern-yellow-bellied racer populations or their critical breeding, foraging, thermal and overwintering habitats within the Milk River and South Saskatchewan River watersheds. These recommendations apply to the full extent of their range within the Grassland Natural Region of Alberta. Further research (Section 8) is required to gain a better understanding of the ecology and population dynamics of these reptiles as well as to better understand the influence grazing has on these species, their critical habitats and their primary prey.

8.1 General Recommendations

Native Prairie Conservation

- Retain native prairie and avoid cultivation or development along major river valleys, coulees or drainages and associated uplands where prairie rattlesnakes, bullsnakes, short-horned lizards and eastern yellow-bellied racers occur (Kissner 2004, Kissner and Nicholson 2003b, Taylor 2004, Martino *et al.* 2012).

- Avoid the conversion of native prairie to cropland. Natural environments provide important habitat for the majority of reptiles and amphibians in Alberta (Cottonwood Consultants 1986). Cultivation of native grasslands is considered a major threat to prairie rattlesnakes, bullsnakes and short-horned lizards (Cottonwood Consultants 1986, Taylor 2004, James pers. comm.). Cultivation can result in direct mortality of these species (*e.g.*, through fatal contact with agricultural equipment; Jørgensen 2009). Tall and dense cropland or tame pasture vegetation may also render areas unsuitable to bullsnakes, prairie rattlesnakes and short-horned lizards by impeding their movement and dispersal across the landscape. Cropland conversion is likely to negatively affect racers as well due to the loss of shrubs for shelter (Parks Canada Agency 2010). Retaining cropland in cities and towns may provide some marginal benefit to prairie rattlesnakes in comparison to urban development by providing movement corridors, a degree of protective cover and some resource opportunities (Andrus 2010).
- Maintain low to moderate shrub densities (in particular sagebrush flats) in native prairie along river valleys, coulees or drainages in prairie rattlesnake, bullsnake, short-horned lizard and eastern yellow-bellied racer habitats. Shrub cover (in particular sagebrush cover) is an important component of prairie rattlesnake, bullsnake and racer foraging habitats (Didiuk 1999, Kissner 2004, Martino *et al.* 2012). Silver sagebrush vegetation also aids in thermoregulation and provides shelter for snakes and short-horned lizards.

Pest Control

- Avoid the use of insecticides and other pesticides in cropland within the vicinity of short-horned lizard, bullsnake, prairie rattlesnake and eastern yellow-bellied racer habitats. These products can reduce the availability of insect or rodent prey or result in bioaccumulation of toxins in prey species (FWD 1990, Cotterill 1997). The use of insecticides may be especially harmful to short-horned lizards and eastern yellow-bellied racers as their diets consist mainly of insects such as ants, grasshoppers, crickets and beetles (James *et al.* 1997; Parks Canada Agency 2010).
- Discourage the use of poisoning programs to control high densities of Richardson's ground squirrels or northern pocket gophers. Rodent poisoning reduces the available prey base for prairie rattlesnakes and bullsnakes and may result in indirect poisoning of these reptiles. Small mammals are not only an important food source, but their burrowing activities provide important snake refugia (Gannon and Secoy 1985, Chizar *et al.* 1996). Encourage alternative ground squirrel population control strategies such as promoting snakes and other natural predators as biological control agents. Investigate vegetation management techniques, such as retaining greater amounts of taller and denser grassland patches to manage ground squirrel populations at appropriate densities (Michener and Schmutz 2002).
- Maintain viable American badger populations in prairie rattlesnake, bullsnake and yellow-bellied racer habitats (Didiuk 1999, Fast and Gates 2003; Martino *et al.* 2012). Badgers play an important role in either creating suitable snake dens or keeping burrow entrances of hibernacula open where slopes are unstable (Didiuk 1999). Badger burrows also appear to be a critical component of gravid female prairie rattlesnake habitat (Fast 2003). Along with ground squirrels, badgers create burrows that are critically important for the survival of eastern yellow-bellied racers (Martino *et al.* 2012).

Habitat Protection and Industrial Development Mitigation

- Avoid disturbance (from human activities or developments) to sparsely vegetated south-, southeast- or east-facing slopes and slump areas along river valleys, coulees and major drainages at any time of the year (Didiuk 1999, James 2002). These areas provide critical overwintering habitat to short-horned lizards, prairie rattlesnakes, bullsnakes and eastern yellow-bellied racers.
- Use Protective Notations under the *Public Lands Act* to protect key reptile habitat, such as has been done for short-horned lizards in the Manyberries badlands area.
- Protect known and high potential bullsnake, prairie rattlesnake and eastern yellow-bellied racer hibernacula sites from disturbance and development. Continued availability of "safe" hibernacula is considered the most critical limiting factor for snakes in Alberta (Cottonwood Consultants 1986). Large aggregations of snakes in these dens are extremely vulnerable to disturbance and "catastrophic events" such as human persecution or flooding (Didiuk 1999). Destruction of hibernacula can therefore have a significant impact on local and regional populations (Didiuk 1999). Snakes occupy hibernacula from early October to late April each year. At C.F.B. Suffield, prairie rattlesnakes and bullsnakes had prolonged periods of remaining near hibernacula in the spring and fall, increasing their vulnerability to human disturbance of these sites.
- Abide by set-back distances and timing restrictions recommended by Alberta Environment and Parks (AEP), Fish and Wildlife Division for high, medium and low impact human activities, including industrial development, near prairie rattlesnake and bullsnake hibernacula and suitable short-horned lizard habitat (GoA 2011). AEP recommends a 200 m, year-round set-back for short- and long-term vegetation disturbance activities from prairie rattlesnake and bullsnake hibernacula. This includes seismic activities and wellsite, powerline, pipeline, battery and road construction. Bullsnake and prairie rattlesnake rookeries have a 200 m set-back from March 15th to October 31st and a 50 m to 200 m set-back (depending on level of disturbance) between November 1st and March 14th. Low disturbance and high disturbance activities are restricted within 100 m and 200 m, respectively, of short-horned lizard preferred habitat.
- Known hibernacula sites in Alberta are listed in the provincial Fish and Wildlife Management Information System (FWMIS). Landowners should be encouraged to contribute information to this database.
- The HSI and RSF maps developed for the short-horned lizard and prairie rattlesnake should be used to inform environmental planning of oil and gas activities to ensure high potential key habitats are avoided. Once additional information is available, HSI or RSF models should be developed for bullsnakes and yellow-bellied racers.
- Exact locations of snake hibernacula and critical short-horned lizard habitat are considered highly sensitive data and its use should be carefully managed (Didiuk 1999, James 2002).
- In areas with suitable prairie rattlesnake, bullsnake or racer habitat, oil and gas companies should routinely monitor and remove snakes from pipeline trenches (Didiuk 1999).
- Given the extreme vulnerability of short-horned lizard populations to disturbance events, and the sensitivity of the habitats they occupy to erosion damage, oil and gas and other resource developments should be avoided in critical short-horned lizard habitats (James

2002), particularly the Manyberries badlands area (SRD 2004). Oil and gas and other resource developments not only disrupt sensitive habitat, but can also potentially open-up lizard habitats to recreational users and increased traffic mortality.

- If oil and gas activity must occur in an area populated by short-horned lizards, James (2002) suggests that activity should occur away from coulee rims and on more densely vegetated, north-facing slopes (if it is necessary to traverse slopes). If short-horned lizards are found within an area of activity, James (2002) recommends that the lizards be moved at least 100 m away from the area of activity.
- Conduct pre-development wildlife surveys in areas of suitable habitat to locate potential prairie rattlesnake, bullsnake and yellow-bellied racer hibernacula and short-horned lizard populations. To search an area with high potential to harbour short-horned lizards, survey protocol guidelines outlined by James (2002) should be observed. These include using experienced or trained field biologists; conducting surveys in late July or early August (when young are born); prioritizing search efforts in high potential habitats; allocating sufficient time to search efforts; and ensuring searching is conducted in appropriate weather conditions (GoA 2013).
- Oil and gas activities in prairie rattlesnake, bullsnake, racer and short-horned lizard habitats should observe methods to minimize disturbance to native prairie and ensure appropriate reclamation.

Road Mortality

Suggested methods to reduce road-kill of snakes include (Didiuk 1999, Rose 2001):

- Avoid development of paved roads near and parallel to major river valleys in the vicinity of known or potential hibernacula;
- Re-direct traffic away from existing roads along valleys near known or potential hibernacula;
- Where possible, schedule industrial development and associated traffic before May, or from mid-June to mid-August and after September near known or potential hibernacula;
- Night-time traffic will have less impact as snakes move less at night and only when it is warm enough;
- Implement traffic speed controls and snake crossing signs in areas with high densities of snakes and along key dispersal corridors;
- In high mortality zones, implement snake and small mammal underpasses, where possible.
- Short-horned lizards are also highly susceptible to road mortality as lizards remain motionless as a defense mechanism and their small size and camouflage make them very difficult to detect (James *et al.* 1997). To minimize possible road mortality of short-horned lizards, avoid or limit road development, recreational motorized vehicle activity and industry traffic along the coulees and river valleys in high potential short-horned lizard habitats, particularly along the slopes and rims. Because male short-horned lizards may also venture out into native grassland beyond the rims of valleys and coulees (Powell and Russell 1998), recreational activity and road traffic should also include set-backs from these sites.

Flooding

- Avoid the creation of reservoirs or dams along major rivers in southeastern Alberta to avoid flooding prairie rattlesnake, bullsnake or yellow-bellied racer hibernacula and critical short-horned lizard habitat (James *et al.* 1997, Didiuk 1999).

Public Awareness and Intentional Persecution

- Intentional persecution including killing of snakes and den vandalism are considered significant limiting factors for bullsnakes and prairie rattlesnakes (Watson and Russell 1997, Kissner and Nicholson 2003b). Public awareness and education campaigns are thought to be an important means to improve the public perception of snakes and inform people about how to avoid or appropriately deal with snake encounters (Kissner and Nicholson 2003b). Public outreach programs could be modeled on initiatives such as the Lethbridge Rattlesnake Conservation Project (Ernst 2002). Teaching the public about the biology and natural history of snakes can increase appreciation for snakes. It is also important to promote the benefits of bullsnakes and prairie rattlesnakes as small mammal control agents (Diller and Johnson 1988, Didiuk 1999).
- Short-horned lizards, as a curiosity item, may face a greater threat from collectors than from intentional persecutors. As an Endangered species, short-horned lizards are fully protected under the *Wildlife Act*. It is illegal to kill, harm, possess, buy or sell these lizards. Because short-horned lizards also occur in urban areas, including the City of Medicine Hat and the Town of Redcliff, they may also be preyed upon by domestic pets (Powell and Russell 1998). Initiatives are needed to inform the public and industry about the vulnerable nature of short-horned lizard populations and their sensitivity to disturbance, particularly in Medicine Hat and Redcliff.

8.2 Grazing Recommendations

Grazing systems that will benefit bullsnakes, prairie rattlesnakes, yellow-bellied racers and short-horned lizards should aim to:

- Maintain the integrity of, or limit impact to, key habitat components (*i.e.*, hibernacula, nesting sites or gravid female habitat); and
- Enhance the suitability of summer foraging sites by creating a patchy landscape with moderate shrub cover (*i.e.*, sagebrush), open basking sites and a diversity of cover types to promote the diversity, abundance and long-term stability of small mammal and invertebrate prey populations.

Although prairie rattlesnakes are known to travel up to 25 km, the majority of rattlesnakes and bullsnakes occur within 15 km of hibernacula areas in river valleys or associated coulees (Didiuk 1999, Nicholson pers. comm.). Grazing in uplands within this 15 km zone is most likely to influence the health, integrity or sustainability of prairie rattlesnake and bullsnake foraging sites (Kissner 2004, Nicholson pers. comm.). Grazing within 100 m of valleys (including coulees and valley breaks and bottoms) has potential to impact short-horned lizard habitats (Taylor 2004). Grazing within 1 km to 5 km of rivers, coulees and drainages has greatest potential to impact bullsnake, prairie rattlesnake and eastern yellow-bellied racer hibernacula, nesting sites or gravid

female habitat (Kissner 2004, Kissner and Nicholson 2003b). Riparian area grazing is unlikely to have a significant impact on short-horned lizards, but may impact foraging opportunities for bullsnakes (Kissner and Nicholson 2003b) and summer habitat of prairie rattlesnake and eastern yellow-bellied racers (Martino *et al.* 2012; Gardiner *et al.* 2013).

Final decisions regarding the type of grazing system that is most appropriate and calculation of suitable stocking rates that would benefit prairie rattlesnakes, bullsnakes, eastern yellow-bellied racers and short-horned lizards need to be made on a case-by-case basis. Local conditions need to be considered, including vegetation type, range and riparian health, and the distribution and location of valued habitat components across the landscape. No one grazing system can be universally applied, and each must be tailored to local environmental conditions. Control and flexibility over stocking rates and dispersion of livestock use are two key properties of an optimum grazing system. Appropriate grazing systems should be developed site-specifically for participating ranches as part of the Multiple Species at Risk (MULTISAR) conservation program.

The following grazing recommendations discuss key management principles that will benefit bullsnakes, prairie rattlesnakes, eastern yellow-bellied racers and short-horned lizards based on available knowledge of their ecology and habitat requirements.

- Prevent cattle-induced slope erosion or trampling along south-facing slopes near known or high potential bullsnake, prairie rattlesnake or eastern yellow-bellied racer hibernacula sites (Didiuk 1999). Monitor the proximity of cattle access trails to these critical over-wintering sites. Where necessary, use fencing to exclude cattle from areas with known hibernacula or areas with high potential to harbour suitable hibernacula (Didiuk 1999). Fencing or salt placement can also be used to redirect cattle along alternate access routes into riparian corridors that will not impact critical overwintering sites.
- Avoid grazing near bullsnake, prairie rattlesnake or eastern yellow-bellied racer hibernacula in the spring (April to June) and fall (September to October) (Kissner pers. comm., Nicholson pers. comm.). Snakes will congregate and bask near hibernacula at these times of the year (Didiuk 1999).
- Maintain essential vegetation cover (*e.g.*, silver sagebrush or dense grass) near hibernacula and gravid female prairie rattlesnake habitat. Assess and monitor the impact of livestock grazing on this vegetation.
- To reduce risks of trampling to young individuals, avoid concentrating cattle use near to known or potential prairie rattlesnake gravid female habitats, bullsnake and racer nesting areas or short-horned lizard birthing sites in the late summer (from August to September).
- Place salt and stock water at least 200 m away from waterbodies, river escarpments, coulee edges and south-facing valley slopes (Kissner pers. comm., Nicholson pers. comm.).
- Avoid spring grazing near slopes and in riparian areas when soils are moist and more susceptible to slumping. This will help to maintain slope stability near hibernacula or bullsnake nesting areas. Avoid summer use of riparian areas in known prairie rattlesnake and eastern yellow-bellied racer territory given these species' preference for this habitat type at that particular time of year (Martino *et al.* 2012; Gardiner *et al.* 2013). Lowland

pastures also appear to serve as important summer habitat for racers and bullsnakes and should therefore receive restricted livestock use during the summer months if possible (Martino *et al.* 2012).

- Avoid continuous heavy livestock grazing in prairie rattlesnake, bullsnake, eastern yellow-bellied racer and short-horned lizard habitats, including upland and riparian sites. Consistent heavy grazing increases trampling risks and is more likely to result in slope damage and erosion that could affect hibernacula. In riparian areas, continuous grazing can reduce plant species diversity and eliminate woody vegetation cover. This may have detrimental impacts to bullsnake and yellow-bellied racer foraging habitats and to thermal cover areas used by prairie rattlesnakes (Didiuk 1999, Kissner and Nicholson 2003b). In upland habitats, continuous heavy grazing can depress populations of most small mammals, reduce diversity of prey species and diminish shelter and thermal vegetative cover that is important to bullsnakes, prairie rattlesnakes and yellow-bellied racers.
- Promote rotational or deferred grazing systems to control the timing of cattle grazing in critical reptile habitats to avoid use near hibernacula or birthing areas in the spring and fall. These grazing systems also allow for improved plant vigour in grasses and increase the likelihood of seed set. Improved plant vigour and seed set benefits cover and foraging opportunities for small mammals and insect prey, including seed-harvester ant species (Fagerstone and Ramey 1996, Nash *et al.* 2000). Rotational or deferred grazing systems are generally recommended to maintain or improve the health of native prairie in the Dry Mixedgrass Natural Subregion, particularly in areas with sandy soil ecological sites.
- Use light to moderate stocking rates to promote patchy grazing and heterogeneous vegetation heights, with areas of heavy, moderate, light and no use. Richardson's ground squirrel habitat and snake basking sites are created in more heavily used areas. Areas with taller vegetation and denser litter cover provide shelter for prairie rattlesnakes, bullsnakes and yellow-bellied racers and also provide suitable cover types to a greater diversity of prey.
- A proper use factor (*i.e.*, total biomass consumption) of 25% to 50% is recommended for sustaining productive native prairie habitat in the Dry Mixedgrass and Mixedgrass Natural Subregions (Adams *et al.* 2013a,b).
- Adjust stocking rates in accordance with range and riparian health. Reduce stocking rates during drought periods.

9 RESEARCH RECOMMENDATIONS

To inform management decisions, additional research is needed to characterize the key habitat needs, foraging requirements, and dispersal patterns of prairie rattlesnakes, bullsnakes, short-horned lizards and eastern yellow-bellied racers. Research is also needed to study the effects of widespread land use activities (*e.g.*, cattle grazing) on these species. Few studies of this kind have been done to date. Recent advances have been made to confirm historical hibernacula locations, locate new hibernacula, inventory populations and develop long-term population monitoring protocols for prairie rattlesnakes and short-horned lizards (Rose 2001, Nicholson and Rose 2001, James 2002, Fast 2003). Comparatively, bullsnakes and yellow-bellied racers have

been poorly researched in Alberta (Kissner and Nicholson 2003b). The provincial road-kill monitoring program for snakes may improve knowledge regarding the distribution of bullsnakes and prairie rattlesnakes in Alberta and their habitat preferences (ESCC 2000).

The goal of grazing management research should be to assess exactly how much of an impact livestock grazing is having on critical reptile habitat components (including hibernacula, nesting or birthing sites, thermal cover and foraging habitat) and to determine which grazing strategies (*i.e.*, timing, distribution, stocking rates) are most beneficial or are most effective at mitigating negative impacts.

Key research needs are described below for each species.

9.1 Rattlesnake

- Continue efforts to conduct long-term population monitoring of prairie rattlesnakes following suggestions provided by Rose (2001). Snake counts at hibernacula are the easiest means to monitor snake populations, but they do not necessarily provide accurate estimates of population sizes. Mark and recapture studies are required in order for population sizes and trends to be accurately measured (Nicholson pers. comm., Kissner pers. comm.).
- Continue efforts to locate and map out prairie rattlesnake hibernacula to protect these sites from development pressures (Nicholson and Rose 2001, Rose 2001, Fast 2003). Remote sensing and GIS-based models have proven useful in the location of new hibernacula (Nicholson and Rose 2001).
- Further investigate the characteristics of vegetation present at den sites (Rose 2001).
- Confirm the characteristics of rookery habitat for gravid female prairie rattlesnakes (Didiuk 1999). This includes determining the movement radius of gravid females from dens and analyzing their habitat selection (Fast and Gates 2003). This information is needed to ensure that recommended set-back distances from hibernacula ensure that gravid females are not disturbed (Fast 2003).
- Continue research to assess the activity patterns of prairie rattlesnakes to better understand its thermal ecology, diet and habitat requirements (Watson and Russell 1997, Didiuk 1999). A research initiative is presently underway to assess dispersal patterns and foraging of prairie rattlesnakes in upland prairies in relation to land use in southeastern Alberta (Jorgensen pers. comm.).
- Information from the snake road-kill monitoring program should be used to guide mitigation efforts to reduce the frequency of road mortality (ESCC 2000). This monitoring program can also help to assess the magnitude of road-induced snake mortality and better understand prairie rattlesnake and bullsnake dispersal patterns and habitat use.
- Research the relative importance of northern pocket gophers and Richardson's ground squirrels and their burrows as a source of food for prairie rattlesnakes (and bullsnakes) and as summer and winter shelter (Didiuk 1999).
- Evaluate the impact of rodent poisoning on prairie rattlesnake and bullsnake populations.

- Assess high potential and known hibernaculum sites for erosion damage and reduced vegetation cover due to cattle grazing (Rose 2001). Assess the role of cattle in causing slumping that creates new hibernacula sites for snakes.
- Assess the impact of cattle grazing on gravid female habitats. Evaluate the frequency and intensity of cattle use in these habitats.
- Evaluate small mammal prey densities in varying land use types, including land managed under different grazing systems or grazing intensities (including ungrazed habitat).

9.2 Bullsnake

- Collect additional baseline data on bullsnake population parameters to establish population sizes and trends in Alberta (Kissner and Nicholson 2003b).
- Research habitat use, dispersal patterns and habitat requirements of the bullsnake (Didiuk 1999, Kissner and Nicholson 2003b). At present there is little information available on specific habitat requirements of this species across its range in Alberta. In particular, little information is available regarding the characteristics of communal bullsnake nesting habitats. Radio-telemetry studies as opposed to simple trapping studies can provide valuable data on the amount of time spent in certain habitats, what activities are carried out in each habitat and provide information on the combination of habitat types that are used by snakes (Kissner and Nicholson 2003b). This technique can also be useful for locating hibernacula and nesting sites.
- Research the effects of grazing on bullsnake nesting sites and hibernacula. The potential for grazing to impact bullsnake upland and riparian area foraging habitats also requires further study.

9.3 Short-horned Lizard

- Continue efforts to monitor short-horned lizard populations and determine population trends as recommended by James (2002), with annual surveys conducted at select locations and less frequent surveys conducted in outlying areas.
- Evaluate the overlap of cattle grazing activities and short-horned lizard distribution.
- Determine the impact of livestock on critical short-horned lizard overwintering habitats.
- Assess the role of grazing in maintaining suitable microsites for short-horned lizards (*i.e.*, maintaining basking sites with reduced forb and grass cover) (Reynolds 1979).
- Assess the potential impact of trampling as a source of mortality among juvenile and adult short-horned lizards.
- Assess the use of cattle trails as dispersal corridors for short-horned lizards in upland or lowland areas.

9.4 Yellow-bellied Racer

- Conduct long-term population monitoring of eastern yellow-bellied racers.
- Make efforts to locate and map racer hibernacula to protect these sites from development pressures.
- Conduct research to determine major characteristics of preferred racer egg-laying habitat.

- Determine the preferred summer foraging habitat characteristics for yellow-bellied racers in Alberta.
- Research the effects of livestock grazing on racer hibernacula sites and summer dispersal habitat.
- Research the major characteristics of racer reproductive biology in Alberta and Saskatchewan (*e.g.*, age of sexual maturity, clutch sizes, egg laying date, incubation time, *etc.*).
- Determine the population size and trend of eastern yellow-bellied racers in Alberta and Saskatchewan.
- Research gene flow between Alberta, Saskatchewan and Montana racer populations.

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B. HERPETILE GROUP 2 (PLAINS HOGNOSE SNAKE, PLAINS SPADEFOOT, GREAT PLAINS TOAD AND CANADIAN TOAD)

1 INTRODUCTION

The purpose of this report is to summarize and compare the ecology and habitat requirements of four herpetofauna (amphibian and reptile) species that occur in similar habitats within the Milk River and South Saskatchewan River watersheds: the plains hognose snake (*Heterodon nasicus*), plains spadefoot (*Spea bombifrons*), Great Plains toad (*Anaxyrus cognatus*) and Canadian toad (*Anaxyrus hemiophrys*). Based on this information, the potential effects of grazing and various grazing systems on these species, their critical habitats and their prey are discussed. This discussion is followed by a summary of recommended beneficial management practices to enhance or protect these species and their habitat in the Milk River and South Saskatchewan River watersheds and throughout their range in southeastern Alberta. Lastly, several research recommendations are presented that would help to better understand these species and their response to management practices.

Plains hognose snakes, plains spadefoot toads and Great Plains toads reach the northern extreme of their North American range in southeastern Alberta, southern Saskatchewan and the extreme southwest corner of Manitoba. These species are all adapted to arid mixedgrass and dry mixedgrass prairie regions with sandy soils. The Canadian toad, in contrast, is found throughout eastern Alberta, much of Saskatchewan and southern Manitoba where it occurs in regions ranging from dry prairie to boreal forest (Hamilton *et al.* 1998). The ability of all four herpetile species to burrow deeply into sandy soils affords them protection from drought and predators, and provides suitable overwintering habitat. Three of these species – plains hognose snake, plains spadefoot toad and Canadian toad – are listed as 'May Be At Risk' of extirpation in Alberta due to their rarity, suspected population declines and their sensitivity to habitat disturbances, while the fourth species – Great Plains toad – is listed as 'Sensitive' (Government of Alberta [GoA] 2016). Plains hognose snake, plains spadefoot toad, Great Plains toad and Canadian toad are also designated as 'non-game animals' under Alberta's *Wildlife Act*, and are therefore protected from hunting or collecting. Plains hognose snake dens are protected from disturbance between September 1 to April 30 under the *Wildlife Act.*

Plains hognose snakes, plains spadefoot toads, Great Plains toads and Canadian toads have not been extensively studied in Alberta and ongoing monitoring is necessary to accurately assess their distribution, population trends, habitat requirements and response to land management practices. Livestock grazing is the dominant land-use within the ranges of these species in the Milk River and South Saskatchewan River watersheds. An understanding of the potential impacts of grazing and associated land use activities on these species and their habitats is therefore important for their management and long-term conservation. It is also important to consider strategies to minimize potential impacts due to other human activities, such as industrial development, which can pose a risk to these species or their habitats.

2 PLAINS HOGNOSE SNAKE

2.1 Background

The plains hognose snake is the only species of *Heterodon* in Alberta. It is a small- to medium-sized, brown- to grey-coloured snake with a sharply upturned rostral scale, no facial pits and three to five rows of darker brown blotches down its dorsal and lateral surfaces. The plains hognose snake was originally considered a subspecies of the western hognose snake, but the three former subspecies of the western hognose snake were recently elevated to their own species status (Crother *et al.* 2012). The species derives its name from a sharply upturned rostral scale (Wright and Didiuk 1998). This species is considered extremely rare in Alberta, with fewer than 100 sites or specimens documented in the province (GoA 2016). The occurrence of plains hognose snakes in Alberta is loosely associated with the major river valleys in the extreme southeastern corner of the province. Some evidence indicates that there are two separate populations – north and south – of this snake in Alberta (Wright and Didiuk 1998). The "south population" occurs in the Milk River Canyon/Wild Horse/Manyberries region and is contiguous with the Montana population of this species (Wright and Didiuk 1998). The "north population" is found along the South Saskatchewan River from Medicine Hat to Empress (Wright and Didiuk 1998). This species also occurs in southwestern Saskatchewan, west of the Missouri Coteau (Wright and Didiuk 1998). A relict population is found in the sandhills of southwestern Manitoba (Leavesley 1987). In the United States, the plains hognose snake occurs from North Dakota east to Minnesota, west to Montana, and south to northern Texas and west central New Mexico (Wright and Didiuk 1998).

The plains hognose snake is listed as 'May Be At Risk' in Alberta under the General Status listing (GoA 2016). The species is designated as a 'non-game animal' under Alberta's *Wildlife Act*, making it illegal to kill, possess, buy or sell hognose snakes. Nationally, the plains hognose snake has not been assessed by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) and is not listed under Schedule 1 of the *Species at Risk Act* (*SARA*) (COSEWIC 2016). The plains hognose snake is scheduled to be assessed by COSEWIC in April 2017 (Davy pers. comm.). Internationally, the plains hognose snake has been designated as a species of 'Least Concern' by the International Union for Conservation of Nature and Natural Resources (IUCN) (Hammerson *et al.* 2007).

More information is needed to determine plains hognose snake population sizes and trends and to accurately map out the range of this species in Alberta. The apparent scarcity of this species may be due to its secretive behavior and misidentification as the bullsnake (*Pituophis catenifer sayi*) or the prairie rattlesnake (*Crotalus viridis*). An increasing number of records in recent years indicate that this species may exist in greater numbers than previously believed (Wright and Didiuk 1998). Although the current population trend is unknown and more research is needed, factors that are thought to be limiting to the plains hognose snake include habitat loss or alteration, industrial and agricultural activities, roads, intentional persecution and the availability of suitable sandy habitat (Wright and Didiuk 1998, GoA 2016).

2.2 Ecology

Plains hognose snakes emerge from hibernation in late April or early May and remain active for approximately 133 days, until mid-September (Pendlebury 1976). Plains hognose snakes re-enter hibernation in late September to early October (Wright and Didiuk 1998). This species has a diurnal activity pattern, and is most active during the mornings and evenings on sunny days when air temperatures range between 10°C to 20°C (Wright and Didiuk 1998). Plains hognose snakes retreat to underground shelters at night and during mid-day (Averill-Murray 2006). Plains hognose snakes will either use rodent burrows as shelters or create their own shelters by digging into loose sandy or loamy soil (Averill-Murray 2006).

The mating season for plains hognose snakes generally occurs in the spring, although mating also occurs in the fall (Platt 1969, Averill-Murray 2006). The egg laying period in Alberta often extends from the second week of June to the last week of July (Wright and Didiuk 1998). During this period females lay between four and 23 eggs, with a mean clutch size of approximately nine eggs (Russell and Bauer 2000). Eggs are usually laid in sandy soil at a depth of approximately 90 mm (Russell and Bauer 2000), although Averill-Murray (2006) suggested that eggs are laid below several centimeters of sandy or loamy soil (Averill-Murray 2006). Clutches are produced every other year, and are incubated for approximately 60 days (Russell and Bauer 2000). Young typically measure 15 cm to 20 cm in length when first hatched (Averill-Murray 2006). Males reach sexual maturity as early as 21 months, whereas females require more than two years to attain breeding age (Werler and Dixon 2000). Adult plains hognose snakes may reach more than 90 cm in length (Averill-Murray 2006). Studies of seasonal activities of plains hognose snakes in Kansas show that the species often exhibits a period of reduced activity in late July or late August (Platt 1969, Platt 1989).

Plains hognose snakes do not constrict their prey, but may use their bodies to help restrain prey (Averill-Murray 2006). To help subdue prey, plains hognose snakes have evolved enlarged rear maxillary fangs (Kroll 1976). The fangs normally rest parallel to the head, pointing to the rear, but extend to a 45° angle when the plains hognose snake opens its mouth to engulf prey. These enlarged teeth provide a strong grip on prey and allow the snake to introduce venom from its salivary glands into the prey's wounds (Averill-Murray 2006). The plains hognose snake also has enlarged adrenal glands which provide physiological resistance to poisons contained in toad skins (Werler and Dixon 2000).

2.2.1 Diet

Toads are considered the main food item for plains hognose snakes (Wright and Didiuk 1998). Other prey items include frogs, lizards, garter snakes, salamanders, tadpoles, lizard eggs, snake eggs, small birds and various small rodents (Munro 1949, Pendlebury 1976, Russell and Bauer 2000). The young of northern pocket gophers (*Thomomys talpoides*) and thirteen-lined ground squirrels (*Spermophilus tridecemlineatus*) are also thought to be important prey items (Wright and Didiuk 1998). At the Canadian Forces Base (C.F.B.) Suffield, plains hognose snakes were captured with the greatest frequency in areas where northern pocket gophers were common. It is not clear whether this snake also favours Richardson's ground squirrel (*Spermophilus richardsonii*) colonies, as this rodent was absent on the study area at C.F.B. Suffield (Wright and Didiuk 1998, Didiuk 1999a).

2.2.2 Predators

Little is known about the effect of predation on plains hognose snake populations in Alberta (Wright and Didiuk 1998). Striped skunks (*Mephitus mephitus*), American badgers (*Taxidea taxus*), coyotes (*Canis latrans*) and red-tailed hawks (*Buteo jamaicensis*) are known predators of snakes or snake eggs that occur within the plains hognose snake range in Alberta (Wright and Didiuk 1998).

2.3 Habitat Requirements

2.3.1 General

The plains hognose snake occurs within the Dry Mixedgrass and Mixedgrass natural subregions of southeastern Alberta (Wright and Didiuk 1998, Russel and Bauer 2000). The plains hognose snake prefers the dry prairies, particularly those with friable, sandy soils and sparse vegetation (Pendlebury 1976, Russell and Bauer 2000). According to Russell and Bauer (2000, p.96) this species "is found in sandy or gravelly areas with low-growing plants, sandhills, dry stream bottoms, often near areas of water in typical badlands country." Recent records from C.F.B. Suffield (1994 to 1997) show that the plains hognose snake occurs in more diverse habitat than formerly believed. Specimens from C.F.B. Suffield were collected on open prairie with either sandy substrate, heavy sod, or near sloughs; on gravelly rolling glacial till plain; open dune sandhill country; and on a riparian sageflat with densely packed soils (Wright and Didiuk 1998). Two road-kill specimens were also collected northeast of Medicine Hat, one along a gravel road bordered by overgrazed rangeland with heavy sod; the second along a major highway bordered by canola fields with a 20 m grassy right-of-way (Wright and Didiuk 1998). Cultivated land is not considered suitable plains hognose snake habitat (Wright and Didiuk 1998). Ditches and rights-of-way along fields and roads likely represent important dispersal corridors for these snakes; however, more research is needed to confirm this hypothesis (Wright and Didiuk 1998).

In Alberta, the plains hognose snake has been found in association with needle-and-thread (*Hesperostipa comata*), sand dock (*Rumex venosus*), prickly rose (*Rosa acicularis*), choke cherry (*Prunus virginiana*), silver sagebrush (*Artemisia cana*), cushion cactus (*Coryphantha vivipara*) and prickly pear cactus (*Opuntia polyacantha*) (Pendlebury 1976).

2.3.2 Hibernacula

Plains hognose snakes either use existing rodent burrows as hibernacula or they dig their own burrows (Wright and Didiuk 1998). Plains hognose snakes have been known to share hibernacula with prairie rattlesnakes, bullsnakes, plains garter snakes (*Thamnophis. radix*) and eastern yellow-bellied racers (*Coluber constrictor flaviventris*) (Wright and Didiuk 1998, Gardiner *et al.* 2011). As previously discussed, prairie rattlesnake hibernacula typically occur along river escarpments, often in stable slump blocks, meander scarps and fissures, subterranean water channels, sinkholes and rocky outcrops (Gannon 1978, Watson and Russell 1997, Didiuk 1999a). Past surveys at C.F.B. Suffield, however, did not document any occurrences of plains hognose snakes sharing hibernacula with other snake species (Wright and Didiuk 1998).

It has been suggested that plains hognose snakes are sedentary and den most often within their summer range in sandy soils or in the burrows of fossorial mammals (Wright and Didiuk 1998). At C.F.B. Suffield, the majority of plains hognose snake captures were further than 5 km from the South Saskatchewan River, suggesting that the river valley does not represent prime overwintering habitat for this species (Wright and Didiuk 1998).
As plains hognose snakes have been frequently captured in Alberta in sandy areas where northern pocket gophers are abundant, it has been speculated that the winter burrows of this rodent provide suitable hibernacula for the plains hognose snake (Wright and Didiuk 1998). As pocket gopher burrows extend below the line of frost-penetration, they offer suitable hibernacula sites, allowing the plains hognose snake to remain on its summer range year-round.

2.3.3 Breeding Habitat

Plains hognose snakes require suitable sandy or friable soil for egg laying (Russell and Bauer 2000). More research is needed to study more specific breeding and egg laying habitat requirements of the plains hognose snake in Alberta.

2.3.4 Foraging Habitat

Native prairie areas with sandy surficial deposits, sufficient vegetation cover and seasonal or permanent wetlands that support important prey populations such as northern pocket gophers, Richardson’s ground squirrels, Great Plains toads and plains spadefoot toads are thought to provide preferred foraging habitats (Wright and Didiuk 1998). More research is needed to study the foraging behaviour and preferred foraging habitats of the plains hognose snake in Alberta.

2.3.5 Area Requirements

Home-range data is not available for plains hognose snakes in Alberta; however, this species is thought to have smaller home ranges than the prairie rattlesnake and bullsnake (Wright and Didiuk 1998). In Kansas, Platt (1969) reported that the longest mean distance traveled by snakes, recaptured after 31 to 50 days, was 200 m. Platt (1969) reported the shortest mean distance traveled was 89 m for snakes recaptured after more than 200 days. Male plains hognose snakes may shift their home range to seek a mate. Snakes may also shift their home ranges to better fulfill their nutritional or cover requirements (Wright and Didiuk 1998). In areas with a stable and sufficient prey base, snakes may travel very little during the active season (Wright and Didiuk 1998). For example, tagged snakes in Minnesota were repeatedly found outside the same burrows of either northern pocket gophers or thirteen-lined ground squirrels (Wright and Didiuk 1998).

3 PLAINS SPADEFOOT

3.1 Background

Plains spadefoots are distinguished from other toad species by their relatively smooth skin, vertical pupils, absent or indistinct parotoid glands and sharp-edged "spades" on each of the hind feet. Plains spadefoots occur in the more arid regions of western North America, generally in association with ephemeral wetlands and sandy soils (Cottonwood Consultants 1986, Klassen

1998, Lauzon 1999). The plains spadefoot occurs primarily in the southeastern portion of Alberta (Lauzon 1999). Historical records indicate that its Alberta range extends from the Montana border north to the village of Veteran and possibly even as far north as the Dillberry Lake/Sounding Lake/Reflex lakes area (Lauzon and Balagus 1998). The western-most extent of the species, based on historical records, is from the Claresholm-Pincher Creek area (Didiuk 2003a). Plains spadefoots have been found as far west as Fort Macleod (Downey 2006, Downey *et al.* 2007). In Canada, the range of the plains spadefoot toad extends east from Alberta into southern Saskatchewan and into the southwest corner of Manitoba (Lauzon 1999). In the United States it occurs south through the plains states to Chihuahua, Mexico and eastern Arizona (Russell and Bauer 2000).

The plains spadefoot toad is designated as 'May Be At Risk' in Alberta under the General Status listing (GoA 2016). The species is not listed specifically under the provincial *Wildlife Act* although it is protected as a 'non-game' animal. Nationally, this species was designated as 'Not at Risk' by COSEWIC in May 2003, and the species remains 'Not at Risk' today (COSEWIC 2016). Plains spadefoots are not listed under Schedule 1 of *SARA*. Internationally, the plains spadefoot toad is listed as a species of 'Least Concern' by the IUCN (IUCN Species Survival Commission [SSC] Amphibian Specialist Group 2015a).

No reliable population numbers exist for the plains spadefoot toads in Alberta (Didiuk 2003a). Although historical trend data is limited and more information is needed to accurately assess plains spadefoot populations in Alberta, their populations appear to be stable (Cottonwood Consultants 1986, Lauzon 1999). However, plains spadefoot population numbers are highly variable from year to year and their range is now larger than previously thought (GoA 2016). According to a 2002 amphibian survey of the Milk River Watershed, plains spadefoots were widely distributed across the watershed, but were not found in the Milk River Ridge area or immediately south of the Cypress Hills (Taylor and Downey 2003). Plains spadefoots were detected at 192 out of 529 stops (36%) surveyed in the Milk River Watershed and 253 breeding sites were present at these stops (Taylor and Downey 2003). Additional surveys conducted in 2005 found plains spadefoot toads at 112 of 642 stops (17%) in the Milk River, Pakowki Lake, and St. Mary River subwatersheds as well as the area west of Fort Macleod (Downey 2006). The 2005 surveys resulted in a slight westward range expansion for the species. A third survey conducted in 2006 found plains spadefoot toads in five different general areas (*i.e.*, Milk River, Writing-on-Stone Provincial Park, west of Pakowki Lake, Fort Macleod and Wrentham/Skiff), including at 50 individual roadside stops (Downey *et al.* 2007).

Declines in amphibian populations world-wide have been attributed to a number of factors, including traffic mortality, pollution (*e.g.*, fertilizers, pesticides, herbicides and acid precipitation), ionizing radiation (UV-B), exotic competitors, predators and pathogens (Canadian Amphibian and Reptile Network [CARCNET] 2002). The most important factor limiting the plains spadefoot appears to be the destruction and alteration of its habitat (Lauzon 1999). Habitat loss and fragmentation have occurred due to extensive cultivation, urbanization, industrial development (*e.g.*, oil and gas exploration), road construction and wetland drainage (Lauzon 1999). Extensive herbicide and pesticide use and water management projects (*e.g.*, dams, reservoirs, and dugouts) are also considered threats to plains spadefoots; however, the impact of these factors requires further study (Klassen 1998, Lauzon 1999).

3.2 Ecology

Plains spadefoot toads are active from late May until the fall, but are rarely seen outside of the breeding period (Russell and Bauer 2000). Plains spadefoots are well-adapted to xeric conditions of deserts and prairies (Lauzon 1999). This species is nocturnal and spends the majority of its time underground, emerging only to breed during favourable conditions or to feed (Lauzon 1999). As their name implies, these toads use large, sharp, spade-shaped tubercles on their hind feet to help them burrow. Plains spadefoots burrow more deeply and emerge less frequently during periods of drought (Bragg 1965).

Plains spadefoots form breeding choruses following heavy rainfall (25 mm to 99 mm) and during periods of warm weather (8°C to 15.5°C) (Didiuk 2003a). Breeding occurs quickly, and may occur more than once in a year if conditions are favourable (Lauzon 1999). Breeding may not occur at all during extended drought periods (Klassen 1998). Didiuk (2003a) postulated that plains spadefoots remain underground in shallow burrows in the spring and summer to allow for foraging while waiting for suitable breeding habitat to become available. In Alberta, breeding has been observed from early May to early July (Klassen 1998, Lauzon 1999). Females lay up to 2,000 eggs in masses of 10 to 250 eggs (Bragg 1965, Collins 1982). Normal development of larvae requires water temperatures within limits of approximately 13°C and 32°C (Blair *et al.* 1995). Under suitable conditions, plains spadefoot eggs hatch rapidly within about two days (Bragg 1965). In Alberta, tadpoles metamorphose 21 to 34 days after hatching, with some requiring up to 60 days (Klassen 1998).

3.2.1 Diet

Adult plains spadefoot toads are carnivorous, and prey on a variety of invertebrates, particularly nocturnal forms such as moths, ants and beetles, as well as spiders and crickets (Bragg 1957, Whitaker *et al.* 1977, Russell and Bauer 2000). Punzo (1991) observed that 82% to 94% of the spring diets of spadefoot toads (*Scaphiopus couchi* and *Spea multiplicata*) in Texas were comprised of beetles, ants, spiders, orthopterans and termites. Adult spadefoot toads may ignore smaller prey items if larger prey is available (Bragg 1957). Spadefoot larvae feed on plankton and organic detritus (Russell and Bauer 2000). Larvae of plains spadefoot toads may also prey on the larvae of other anuran species, including Great Plains toads, other species of spadefoot toads and possibly tiger salamanders (Bragg 1964). Plains spadefoot toad larvae have also shown evidence of cannibalism (Bragg 1964).

3.2.2 Predators

Several snake species prey on adult plains spadefoots including garter snakes, bullsnakes and plains hognose snakes. Tadpoles may be preyed upon by aquatic arthropods (Russell and Bauer 2000).

3.3 Habitat Requirements

3.3.1 General

In Alberta, the plains spadefoot toad occurs primarily in the Grassland Natural Region where it has been observed in dry mixedgrass, mixedgrass and fescue prairie, sagebrush, sand dunes,

floodplains and sandy stream channels (Cottonwood Consultants 1986, Lauzon 1999). Lauzon and Balagus (1998) observed plains spadefoot toads in sandy soils in native grassland and native grassland/shrubland complexes in the Medicine Hat region as well as in native grassland, native grassland/shrubland complexes and tame pasture/hay land in the area near the villages of Youngstown and Veteran. This species is generally found near temporary bodies of water and its distribution is strongly correlated with sandy or friable soils suitable for burrowing (Lauzon 1999, Russell and Bauer 2000). Adults rest in rodent burrows or self-constructed burrows in sand or other easily-penetrated soil (Lauzon 1999).

Cropland is thought to provide unsuitable habitat for plains spadefoot toads due to the alteration of breeding ponds and loss of optimal foraging habitat (Butler and Roberts 1987, Didiuk 2003a). Nevertheless, Downey (2006) observed plains spadefoot toads in cultivated fields and ditches in southern Alberta. However, the persistence of this population in a cultivated area is unknown. Gray and Smith (2005) found that post-metamorphic plains spadefoot toads exhibited smaller body sizes in wetlands surrounded by cropland compared to individuals in wetlands surrounded by native grassland. Smaller body sizes in amphibians are correlated with reduced probability of reproduction and survival (Goater 1994, Scott 1994). Therefore, cultivated areas may represent sink habitats.

As part of the Multiple Species at Risk (MULTISAR) conservation program, a broad-scale habitat suitability index (HSI) model was developed for the plains spadefoot toad in the Milk River Watershed (Taylor 2004a). This model incorporated two equally-weighted variables: soil texture and native prairie class. According to this model, Class 1 native prairie habitats with moderately coarse and coarse soil textures represent preferable plains spadefoot habitat. Class 1 native prairie is comprised of more than 75% native prairie components by quarter-section. Moderately coarse and coarse textured soils include sandy loam, sands and loamy sands.

As a continuation of the MULTISAR HSI work, Resource Selection Function (RSF) models were developed and tested to predict plains spadefoot toad habitat use in southern Alberta (Skilnick and Dodd 2011). This modelling exercise tested numerous biophysical variables derived from such datasets as the Grassland Vegetation Inventory (GVI), Agricultural Region of Alberta Soil Inventory Database (AGRASID), quaternary geology and a digital elevation model (DEM) from which was derived slope, elevation and aspect. Variables selected for the best-fit model included: elevation, soil texture, distance to Sands and Sandy ecological sites, distance to Lentic – Temporary and Lentic – Semi-permanent/Permanent wetlands, distance to irrigated fields (cropland and tame pasture), distance to creeks (perennial and ephemeral), distance to Blowouts ecological sites, distance to eolian deposits and distance to non-irrigated cropland and tame pasture. In the top model for the plains spadefoot toad, relative probability of occurrence increased with proximity to Sand, Sandy and Blowouts ecological sites, irrigated fields, creeks, temporary wetlands and non-irrigated cropland. Relative probability of occurrence was higher with further distances from semi-permanent/permanent wetlands, non-irrigated tame pasture and eolian deposits. Plains spadefoots appeared to use all other soil texture types more than moderately textured soil types. However, there seemed to be a general selection for coarser soils types.

Gray *et al.* (2004a, b) studied landscape structure on amphibian species assemblages in the Southern High Plains of the central United States. The authors found that plains spadefoot toads were positively correlated with increasing wetland density (*i.e.*, decreasing distance between wetlands) and increasing landscape complexity between wetlands. Landscape complexity was related to agricultural activity. The authors speculate that the reason plains spadefoot toads were correlated with high wetland density within agricultural landscapes was that their small body size prevented them from dispersing through these anthropogenically-altered landscapes, resulting in increased abundance near their natal wetlands. Plains spadefoot toads inhabiting wetlands in cultivated landscapes have less diverse diets and smaller postmetamorphic body sizes than toads in wetlands surrounded by native grassland (Gray and Smith 2004, Smith *et al.* 2004). Wetlands in cultivated landscapes may therefore function as habitat sinks as opposed to sources for plains spadefoot toads (Gray 2002).

3.3.2 Hibernacula

To avoid freezing and desiccation during the winter, plains spadefoots burrow deeply into sandy soils, usually to depths below the frost line (Lauzon 1999). These toads are known to burrow up to 1 m deep in order to find a damp hibernation site (Russell and Bauer 2000). Plains spadefoots are also capable of super-cooling to -4.3^{o}C, which helps them to avoid freezing in shallower burrows over the winter (Swanson and Graves 1995).

3.3.3 Breeding Habitat

Ephemeral and temporary wetlands are the preferred breeding habitats of the plains spadefoot toad (Klassen 1998, Lauzon and Balagus 1998, Didiuk 2003a). Breeding opportunities are dependent on the availability of suitable breeding sites, including temporary pools or flooded areas that vary from several centimeters to more than one meter, often with a mud bottom and with either clear or turbid water (Blair *et al.* 1995). Cottonwood Consultants (1986, p.15) reported that plains spadefoots breed in “shallow water of vernal pools on uplands and along streams, semi-permanent ponds, oxbow lakes and stream meander channels”. Seasonal wetlands and the shallow margins of permanent wetlands and lakes may also provide suitable breeding habitat (Didiuk 2003a). Eggs are usually deposited in areas of still, shallow water (Krupa 1994). To ensure survival, water needs to persist until larvae metamorphose.

Taylor and Downey (2003) reported that general conditions of plains spadefoot toad breeding sites in the Milk River Watershed were ephemeral wetlands less than 50 cm deep with little to no aquatic vegetation (Taylor and Downey 2003). Klassen (1998) reported that plains spadefoot toads in the Milk River area bred in sloughs with little vegetation, marshy depressions, flooded cultivated fields, temporary wetlands in pastures, small pools, river backwaters and ditches. Although plains spadefoots appear to be opportunistic in their selection of breeding ponds, Taylor and Downey (2003) observed that the only sites that allowed complete development of tadpoles were ephemeral ponds within native prairie. Ditches and sites near cultivation often fail to retain sufficient water to allow for completion of metamorphosis (Taylor and Downey 2003). Breeding ponds located in native prairie grasslands in the Milk River Watershed occurred in areas dominated by blue grama (*Bouteloua gracilis*), June grass (*Koeleria macrantha*), needle-and-thread, and northern wheatgrass (*Elymus lanceolatus*) (Klassen 1998).

3.3.4 Foraging Habitat

Little is known about the preferred foraging habitats of plains spadefoots in Alberta. Plains spadefoot larvae feed in still, shallow water in flooded areas and temporary, often mud-bottomed pools (Lauzon 1999). Adult plains spadefoots feed above-ground during the night, usually in humid or rainy weather, likely in the vicinity of suitable burrowing habitat.

3.3.5 Area Requirements

More research is needed to determine the summer movements and home range size of plains spadefoot toads (Lauzon 1999). Plains spadefoots are capable of migrating at least 1.6 km to breeding sites (Landreth and Christensen 1971). In Alberta, Klassen (1998) recorded juvenile plains spadefoots more than 2 km from known breeding wetlands; however, these toads may have originated from unknown sites.

4 GREAT PLAINS TOAD

4.1 Background

The Great Plains toad is one of three species of the genus *Anaxyrus* in Alberta, along with *Anaxyrus hemiophrys* (Canadian toad) and *Anaxyrus boreas* (western toad). Great Plains toads are distinguished from other toad species by their relatively large, 'L'-shaped cranial ridges behind the eyes and dark paired blotches with light borders on a grey, light brown or olive-coloured back. Great Plains toads, like plains spadefoot and plains hognose snakes, are restricted to the southeastern corner of Alberta and are at the northern extreme of their range in the province (James 1998, Russell and Bauer 2000). These toads are considered to have a rare and localized distribution in Alberta (Cottonwood Consultants 1986). Their general range in the province extends from the Red Deer River south to the Montana border, and east from town of Taber to the Saskatchewan border (James 1998). Four general regions of Great Plains toad records are present in Alberta: the Middle Sand Hills/C.F.B. Suffield, Tilley/Lake Newell/Vauxhall/Taber, Onefour and Wrentham/Skiff areas (COSEWIC 2010). Great Plains toads also occur in southern Saskatchewan and the extreme southwestern corner of Manitoba (James 1998). A paucity of information is available regarding the historic range of this species in Canada (James 1998). Outside of Canada, Great Plains toads are widely distributed in western North America and the northern half of Mexico (Didiuk 1999).

Great Plains toads are designated as 'Sensitive' in Alberta under the General Status listing (GoA 2016). This species has also been designated a species of 'Special Concern' under the Alberta *Wildlife Act*, meaning it is particularly sensitive to human activities and/or natural disasters. Great Plains toads are protected from harm as a 'non-game animal' under the *Wildlife Act*. COSEWIC designated this species as 'Special Concern' in 1999, and this status was reconfirmed in 2002 and 2010 (COSEWIC 2002, 2016). It remains a species of 'Special Concern' today, and it is listed as such under Schedule 1 of *SARA* (COSEWIC 2016). The Great Plains toad is listed as a species of 'Least Concern' internationally by the IUCN (IUCN SSC Amphibian Specialist Group 2015b).

Three surveys for Great Plains toads have been carried out through the MULTISAR program since the early 2000s (Taylor and Downey 2003, Downey 2006, Downey *et al.* 2007). In 2002, Great Plains toads were observed at 19 out of 529 stops surveyed during a roadside amphibian survey of the Milk River Watershed (Taylor and Downey 2003). At these 19 sites, 21 breeding wetlands were documented, all of which were located east of Lost River (Taylor and Downey 2003). Additional roadside surveys conducted in 2005 did not find any Great Plains toads at 642 stops in the Milk River, Pakowki Lake and St. Mary River subwatersheds as well as the area west of Fort Macleod (Downey 2006). The lack of observations in these regions was assumed to be a result of inadequate precipitation in breeding areas. However, the 2005 surveys did yield the discovery of a new population of Great Plains toads in the area near the hamlets of Wrentham and Skiff. Great Plains toads were heard calling at 14 different wetlands in this new breeding area. In 2006, Great Plains toads were heard calling at two locations (Onefour and Wrentham/Skiff), including at 25 individual stops (Downey *et al.* 2007).

Great Plains toads are thought to be limited by the quality and availability of suitable breeding sites in Alberta (James 1998). Habitat alteration and destruction (*i.e.*, wetland loss or alteration), hydrological changes, cultivation, drought, pollution (*e.g.*, pesticide and herbicide use), road kills and oil and gas exploration and development are considered the main anthropogenic factors that threaten the long-term survival of this species in Alberta (Cottonwood Consultants 1986, Wershler and Smith 1992, James 1998, Didiuk 1999, GoA 2016). Drought, and to a lesser extent predation, are considered to be primary natural limiting factors (Wershler and Smith 1992, James 1998).

As with plains spadefoots and plains hognose snakes, there is limited long-term monitoring data for Great Plains toads, making it difficult to assess their population status (James 1998). Great Plains toad population numbers are thought to fluctuate widely over time given their reliance on rainfall for reproduction (James 1998). Nevertheless, population numbers appear to be declining in Alberta (GoA 2016). Wallis and Wershler (1988) originally estimated the population of Great Plains toad in Alberta to be less than 1,000 adults. Wershler and Smith (1992) estimated the Great Plains toad population in Alberta to be as high as 2,000 adults. Didiuk (1999b) suggested that the total Alberta population could be in the tens of thousands based on findings of substantial numbers of Great Plains toads at C.F.B. Suffield. Pearson (2009) suggested that the population in Alberta between 2,100 and 10,000 individuals. Although it did not provide a specific number, COSEWIC (2010) suggested that the population of Great Plains toads in the province is undoubtedly higher than that predicted in the late 1980s and early 1990s.

4.2 Ecology

Like plains spadefoots, Great Plains toads are opportunistic spring breeders and in dry years will only breed following large precipitation events (Krupa 1994). Great Plains toads may not attempt to breed during years with prolonged drought (James 1998). In Alberta, Great Plains toads usually breed from late April to mid-June (Cottonwood Consultants 1986, Didiuk 1999a). Great Plains toads often form large breeding assemblages and lay their eggs in a communal mass, often in the same location each year (Krupa 1994, James 1998). Females produce approximately 1,300 to 45,000 eggs, with a documented average clutch size of approximately 9,400 eggs (Krupa 1994). Larger females will lay more eggs than smaller females. Great Plains toads are capable of

laying multiple clutches. Eggs will hatch in one to five days depending on the ambient temperature (Krupa 1994). Tadpole metamorphosis can range from 18 to 49 days and is also temperature dependent (Krupa 1994). Desiccation of breeding pools prior to completion of metamorphosis can lead to a high rate of tadpole mortality (Krupa 1994). Newly metamorphosed young have been reported in Alberta as early as June 28 and July 18 (Wershler and Smith 1992). New Great Plains toads will stay in their natal pond area for a month or until it desiccates (Smith and Bragg 1949). Great Plains toads attain sexual maturity in three to five years (Russell and Bauer 2000).

Great Plains toads have numerous adaptations to enable them to survive in arid northern grasslands. One of their primary means of coping with wide temperature fluctuations and prolonged drought is their ability to burrow underground for lengthy periods of time (James 1998, Russell and Bauer 2000). Great Plains toads are dehydration-tolerant, but not freeze-tolerant (Edwards *et al.* 2004). Great Plains toads have adapted to the cold winter temperatures of southern Canada by burrowing underground to below the frost line (up to 74 cm to 104 cm) during fall and winter (Graves and Krupa 2005). Great Plains toads are also mostly nocturnal, foraging mainly at night or during the day during wet periods (James 1998). Large quantities of food are consumed during favourable periods, allowing this toad to burrow and remain dormant during hot, dry weather (Bragg and Smith 1943, James 1998). As Great Plains toads may live as long as two decades, their longevity has been suggested as another survival adaptation (James 1998). Their long lifespan enables them to wait out prolonged dry periods between successful breeding events (James 1998).

4.2.1 Diet

Great Plains toads are thought to be opportunistic feeders, as diet composition is determined principally by prey availability rather than selection (Smith and Bragg 1949). Common prey items include ground-dwelling, nocturnal insects such as beetles, ants and spiders (Smith and Bragg 1949). Adult female Great Plains toads consume a wider variety of prey items than males (Bragg and Smith 1949). Smith *et al.* (2004) observed that newly metamorphosized Great Plains toads fed on scarab beetles and formicid ants. These authors also noted that newly emerged Great Plains toads occupying wetlands within native grassland had more varied diets than those individuals occupying wetlands within cultivated fields. Tadpoles feed primarily on algae and decomposing vegetation or invertebrates (Bragg 1940). Tadpoles will also cannibalize other dead tadpoles (Bragg 1940).

4.2.2 Predators

In Alberta, predators of Great Plains toads include plains hognose snakes and corvids (such as American crows [*Corvus* brachyrhynchos]) (Wershler and Smith 1992, James 1998). Other potential predators of juvenile and adult Great Plains toads include: Swainson's hawks (*Buteo swainsoni*), garter snakes (*Thamnophis* spp.), black billed magpies (*Pica pica*), great blue herons (*Ardea herodias*) and black-crowned night-herons (*Nycticorax nycticorax*) (Bragg 1940, Didiuk 1999b). Great Plains toad tadpoles commonly fall prey to birds, diving beetles (*Hydrophilus triangularis*), insect larvae and plains spadefoot tadpoles (Bragg 1940, James 1998).

4.3 Habitat Requirements

4.3.1 General

In Canada, Great Plains toads are exclusively associated with the native grasslands of the southern Prairie Provinces (COSEWIC 2010). Great Plains toads inhabit sand plain and sandhill habitat types within the Dry Mixedgrass and Mixedgrass Natural Subregions of southeastern Alberta (James 1998, NRC 2006). This toad species typically avoids woodlands and the floodplains of streams (Bragg 1940). All ponds surveyed for Great Plains toad during 1987 to 1990 in Alberta by Wershler and Smith (1992) were located in native prairie; no ponds occupied by Great Plains toads were found in cultivated areas. However, Downey (2006) and Downey *et al.* (2007) observed Great Plains toads calling from wetlands located in cropland. In addition, Didiuk (1999b) observed metamorphosis of Great Plains toads in wetlands situated within tame pastures in C.F.B. Suffield.

HSI modelling for Great Plains toad habitat use in the Milk River Watershed was completed by MULTISAR in the mid-2000s (Taylor 2004b). Three variables were included in the final HSI model: native prairie class, soil order and soil texture. According to this model, Class 1 native prairie areas with Chernozemic / Solonetzic soil orders and moderately coarse (sandy loam) and coarse (sand and loamy sand) textured soils provide optimal habitat for Great Plains toads. As previously discussed, Class 1 native prairie areas are comprised of greater than 75% native prairie by quarter-section. Solonetzic soil orders are thought to be preferred as their poor drainage characteristics accommodate the formation of ephemeral ponds that last long enough to accommodate breeding. Coarse textured soils are preferred as they provide suitable burrowing habitat.

An RSF model was developed for Great Plains toad habitat preferences in southern Alberta for MULTISAR in 2010 using data sets that were generally more refined than those used for the HSI model (Skilnick *et al.* 2010). This modelling exercise tested numerous biophysical variables derived from such datasets as the GVI, AGRASID, quaternary geology and a DEM from which was derived slope, elevation and aspect. Variables selected for the best-fit model included: soil texture, soil Great Group, elevation, wetlands and irrigation canals. According to this model, Great Plains toads appear to select for closeness to ephemeral wetlands; a high percentage of wetlands in a given area; lower elevations; and further distances from Solonetzic soils and irrigation canals. Coarse textured soils were selected for more than moderately textured soils, while finer soils were less preferred than moderate soils.

At the landscape level, Naugle *et al.* (2005a) emphasized the importance of having high numbers of ephemeral wetlands in order to facilitate population explosions and thereby renew populations depleted by drought. However, the previously discussed study by Gray *et al.* (2004a) on landscape structure and its relationship to amphibian habitat preferences in the Southern High Plains of the central United States found that relative abundance of Great Plains toads was not correlated with density of wetlands (or landscape complexity, essentially cropland). Great Plains toad abundance was negatively associated with presence of plains spadefoots, possibly because the larvae of the latter are stronger competitors (Gray *et al.* 2004a,b).

4.3.2 Hibernacula

Great Plains toads are freeze-intolerant and need to burrow below the frost line in order to survive the winter in Canada (Graves and Krupa 2005). Little is known about Great Plains toad overwintering sites in Canada but they are likely characterized by sandy soil, which is easier for toads to burrow in than clay or silty soils (COSEWIC 2010). Fossorial mammal burrows may provide suitable overwintering habitat for Great Plains toads in Alberta (Bragg 1940, Didiuk pers. comm.). In Minnesota, Great Plains toads overwintered in roadside berms and other sites with good drainage (Ewert 1969). American badger burrows were sometimes used by adult Great Plains toads to facilitate burrowing into road beds (Ewert 1969). The soft soil of ant nests may also be preferred hibernacula (Whitford and Melzer 1976). A similar species, the Canadian toad, is known to hibernate in 'Mima-type' mounds that are thought to be at least partly the result of pocket gopher activity (see discussion below) (Breckenridge and Tester 1961).

4.3.3 Breeding Habitat

Great Plains toads prefer clean, clear, shallow ephemeral pools and ditches filled with spring snowmelt and rainwater for breeding habitat (Bragg 1949, Bragg and Smith 1943). In Alberta, Great Plains toads breed mostly in seasonal (ephemeral) waterbodies such as potholes or prairie depressions that are filled with water during spring runoff and early season rainfall (Wershler and Smith 1992). Some permanent and semi-permanent wetlands may also be used, including springs, irrigation projects and dikes in shallow drainages (Wershler and Smith 1992). More permanent waterbodies or those located in irrigated areas provide stable breeding sites during long periods of drought (Wershler and Smith 1992). However, Great Plains toad tadpoles may have a greater chance of survival in temporary rather than permanent wetlands as studies have found that permanent ponds harbour a greater abundance of aquatic and terrestrial predators (Woodward 1983). At C.F.B. Suffield, Didiuk (1999a) reported that Great Plains toads occurred most often in larger seasonal basins in high relief morainal deposits, commonly in association with plains spadefoots. Naugle *et al.* (2005b) observed Great Plains toads in all classes of wetlands surveyed in South Dakota, including temporary, seasonal, semi-permanent, permanent, tillage, man-made (*e.g.*, stock dams) and riverine wetlands.

Water clarity and temperature are thought to be an important characteristic of breeding sites (Bragg 1940, 1950). Great Plains toads reportedly will not breed in muddy or turbid waters and egg masses might not survive in water contaminated with cattle faeces (Bragg 1940, Bragg and Smith 1943, James 1998). Wershler and Smith (1992, p.3) described the typical breeding habitat of Great Plains toads in Alberta as "shallow ponds with relatively fresh, clear water in areas of sandy soil." Bragg (1950) described the ideal Great Plains toad breeding wetland characteristics as shallow (0.5 m to 1.5 m depth), completely clear to semi-clear, with temperatures between 11°C and 20°C, in pools that dry up at least annually.

In terms of vegetation in preferred breeding habitat, Wershler and Smith (1992) found that tall emergent vegetation (*e.g.*, common cattail [*Typha latifolia*] and bulrushes [*Shoenoplectus* spp.]) was uncommon in breeding ponds, whereas submerged vegetation (*e.g.*, pond weeds such as *Potamogeton* spp.) was abundant in some ponds in non-irrigated areas. In irrigated areas, algal growth in breeding ponds was used as calling perches by males and was thought to provide egg attachment sites and shelter for tadpoles (Wershler and Smith 1992). Taylor and Downey (2003)

observed Great Plains toads in ephemeral ponds approximately 1 m deep with clear water and little to no aquatic vegetation. Ewert (1969) found that Great Plains toads tended to avoid wetlands with common cattail and wetlands in forested habitats.

4.3.4 Foraging Habitat

More research is needed to determine the preferred foraging habitat of Great Plains toads in Alberta. Young Great Plains toads tend to remain near breeding ponds until these ponds evaporate (Smith and Bragg 1949, Bragg 1950). As Great Plains toads mature they become nocturnal and will forage in grassland areas during the night within range of suitable sandy soils which allow for burrowing (Smith and Bragg 1949). In Oklahoma, Great Plains toads have been found to burrow and feed in cultivated fields with suitably friable soils (Smith and Bragg 1949).

4.3.5 Area Requirements

Creuse and Whitford (1976) observed Great Plains toads up to 1,600 m from breeding sites. Ewert (1969) observed young-of-the-year Great Plains toads up to 915 m (3,000 ft) away from the nearest pond in his Minnesota study area. This same author estimated that Great Plains toads had elongated home ranges of 610 m (2,000 ft) in length. Total distances moved by individuals during the active season averaged 615 m, with one individual moving a distance of 808 m (2,650 ft) in a single day (Ewert 1969). The majority of movements by individuals during the early (43%) and late (69%) summer were over short distances (*i.e.*, 0 m to 3 m). Movements following breeding consisted of a series of short-distance movements from breeding areas directly to over-wintering sites (Ewert 1969). Movements were interrupted by brief stops for foraging and with retreats into underground burrows for several days at a time (Ewert 1969).

The home range size and seasonal movements of Great Plains toads have not been well studied in Alberta. Incidental observations from pitfall snake traps used at C.F.B. Suffield, suggest that adult Great Plains toads may move approximately 1 km from known breeding ponds (Didiuk pers. comm.). In the Lost River valley area of the extreme southeastern corner of the province, Great Plains toad young-of-the-year were observed on the valley side slopes within 500 m of a presumed breeding pond in August after the pond had dried up (Taylor 2004b).

5 CANADIAN TOAD

5.1 Background

The Canadian toad is another species of the genus *Anaxyrus* found in Alberta and potentially of concern. The Canadian toad is distinguished from the other two toad species by having cranial crests which are either parallel or which diverge anteriorly (Russell and Bauer 2000). In contrast, the cranial crests of the Great Plains toad diverge anteriorly, and the western toad lacks cranial crests entirely (Russell and Bauer 2000). Unlike the Great Plains toad, plains spadefoot and plains hognose snake, the Canadian toad is found throughout much of Alberta, including in the Grassland, Parkland and Boreal Forest Natural Regions (Hamilton *et al.* 1998, Natural Regions Committee [NRC] 2006). For reasons not currently clear, the distribution of the Canadian toad in Alberta appears to be confined to elevations below 1,000 m above sea level (Hamilton *et al.*

1998). Historic records exist for the species as far north as Fort Smith along the Alberta-Northwest Territories border to as far south as the town of Brooks (Hamilton *et al.* 1998). The range of the Canadian toad extends as far west as Calgary and Rocky Mountain House (Hamilton *et al.* 1998).

The Canadian toad is listed as 'May Be At Risk' under the General Status listing in Alberta (GoA 2016). Due to the dearth of information on population status and trend, the Canadian toad is listed as 'Data Deficient' under the provincial *Wildlife Act* (GoA 2014). Nevertheless, Canadian toads are protected as 'non-game animals' under the *Wildlife Act*, making it illegal to kill, possess, buy or sell individuals of this species. The species was last assessed by COSEWIC in 2003 and found to be 'Not at Risk' nationally (COSEWIC 2016). As a result, the Canadian toad is listed as 'Not at Risk' under Schedule 1 of *SARA*. The Canadian toad is listed as a species of 'Least Concern' by the IUCN (IUCN SSC Amphibian Specialist Group 2015c).

In Alberta, Canadian toads have been identified as a species potentially of concern due to apparent declines in both distribution and population size throughout the province (Hamilton *et al.* 1998). Recent data has confirmed that Canadian toad distribution is patchy throughout its range, and the species appears to be exhibiting dramatic declines, at least in Alberta (GoA 2016, COSEWIC 2016). Declines may be greatest in the Parkland and Boreal Forest Natural Regions (NRC 2006, Browne 2009, GoA 2016). Loss and degradation of native grassland and wetlands, the preferred habitats of Canadian toads in southern and central Alberta, are ongoing threats along with drought, oil and gas exploration and development and agricultural use of pesticides (GoA 2016, COSEWIC 2016). However, despite this apparent decline in this province, the species remains numerous and widely distributed throughout much of its Canadian range (COSEWIC 2016).

5.2 Ecology

Canadian toads overwinter underground, a strategy that likely evolved to survive the harsh winters of northern latitudes. Emergence from underground burrows occurs over a five to six week period in spring (Kelleher and Tester 1969). In northern Alberta, Kuyt (1991) observed Canadian toads emerging from their hibernacula in early May. Tester and Breckenridge (1964) observed peak emergence of Canadian toads in Minnesota to be in mid-May (*i.e.*, May 12th to 18th), with first emergence occurring between late April and early May. Tarnsitt (1962) noted emergence as early as March 28th in southern Manitoba during an unusually warm summer. The time of first emergence and the peak of emergence appear to be dependent upon rising temperatures and precipitation and depth of frost in the ground (Tester and Breckenridge 1964, Kelleher and Tester 1969). Adults tend to emerge earlier than immatures (Tester and Breckenridge 1964). Male Canadian toads tend emerge before females (Kelleher and Tester 1969).

Following emergence, males form breeding choruses along the edges of ponds, lakes and wetlands (Tester and Breckenridge 1964). Males continue to call for two months (Breckenridge and Tester 1961). Females arrive at the breeding ponds singly and only stay long enough to mate and lay their eggs (Tester and Breckenridge 1964). Spawning occurs in late May to mid-June in Alberta (Robert and Lewin 1979). Eggs are laid in long strings that may or may not be attached

to submerged vegetation (Roberts and Lewin 1979). Canadian toads appear to be quite prolific compared to frogs in Alberta, with females producing between 3,354 and 5,842 eggs (Roberts and Lewin 1979). High egg production may be a strategy to compensate for the variable breeding habitats used by Canadian toads (Roberts and Lewin 1979).

In northeastern Alberta, metamorphosis occurs from late June to mid-July (Robert and Lewin 1979). Similar dates were recorded for Minnesota (Breckenridge and Tester 1961) and Manitoba (Tarnsitt 1962). Once developed, young-of-the-year can be found along the margins of breeding ponds (Roberts and Lewin 1979). Most Canadian toads return to their hibernacula sites in late August to early September, although some return as late as mid-October (Breckenridge and Tester 1961). Kuyt (1991) observed Canadian toads returning to their overwintering sites in early September in northeastern Alberta. Adults appear to begin hibernation earlier than juveniles (Breckenridge and Tester 1961).

Unlike frog species such as boreal chorus frogs (*Pseudacris maculata*) and wood frogs (*Lithobates sylvatica*), the American toad (*Bufo americanus*), a species closely related to the Canadian toad, lacks cryoprotectants (*i.e.*, freeze-protecting chemicals) and therefore must burrow beneath the frost line in order to survive at northern latitudes (Storey and Storey 1986). It seems reasonable to conclude that Canadian toads are similar to American toads in this regard given their close life history. Canadian toads can burrow more than 1 m below the soil surface (Breckenridge and Tester 1961). Canadian toads also appear to remain active during the winter, burrowing deeper if winter temperatures drop (Breckenridge and Tester 1961).

Breckenridge and Tester (1961) and Tester and Breckenridge (1964) suggested that Canadian toads in Minnesota reached maturity at around 24 months of age (when they are approximately 4.5 cm in length), but they did not breed until the start of their third summer of life. Similarly, Tarnsett (1962) suggested that Canadian toads reach the breeding stage at 2 years of age. However, an analysis of museum specimens from Alberta suggested that male Canadian toads reach maturity and start breeding at one year of age and females reach sexual maturity at two years of age (Eaton *et al.* 2005). Canadian toads in Alberta have been found to live as long as 12 years (Eaton *et al.* 2005). Adult Canadian toad sex ratios ranged from 33% to 38% males (Kelleher and Tester 1969).

In one study, mortality rates for Canadian toads in Minnesota averaged 32.7% for adult males, 31.5% for adult females and 35.9% for juveniles (Kelleher and Tester 1969). Mortality rates are higher for juveniles than for adults due the tendency of the former to disperse to upland areas during summer, making them more vulnerable to predation and climatic factors *(e.g.*, drying out in hot sunlight) (Tester and Breckenridge 1964). In addition, juveniles may not be able to burrow as deep as adults, making them more vulnerable to mortality during harsh winters with deep frost lines (Tester and Breckenridge 1964).

5.2.1 Diet

Canadian toads feed almost exclusively on insects, primarily arthropods (Moore and Strickland 1954). Insects from the Coleoptera (Beetles) and Hymenoptera (Ants, Bees, Wasps and Sawflies) families are most commonly eaten by adults (Moore and Strickland 1954). Non-insect food

consists primarily of spiders (Moore and Strickland 1954). Snails may also be consumed by adult Canadian toads (Moore and Strickland 1954). Juvenile Canadian toads feed primarily on small beetles and flies, with mites, springtails and hymenopterans being of secondary importance (Moore and Strickland 1954). In Manitoba, Canadian toads were observed to feed on dipterans (Flies and Mosquitoes), especially chronomids (non-biting midges) (Tarnsitt 1962).

5.2.2 Predators

Tester and Breckenridge (1964) observed predation on Canadian toads in Minnesota by the plains garter snake (*Thamnophis radix*) and red-tailed hawks. These authors also suspected that raccoons (*Procyon lotor*) and American badgers also preyed on Canadian toads. Tarnsett (1962) observed predation on a Canadian toad by a plains garter snake in southern Manitoba. He included raccoons, red foxes (*Vulpes vulpes*), martens (*Martes americana*), striped skunks, ground squirrels and shrews (*Sorex* spp. and *Blarina brevicauda*) as potential predators of Canadian toads. Other predators of Canadian toads include: American kestrel (*Falco sparverius*), Swainson's hawk (*Buteo swainsoni*), red-tailed hawk, common crows (*Corvus brachyrhynchos*), great blue heron and black-crowned night-heron (summarized by Didiuk 2003b). Canadian toad larvae may be preyed upon by other amphibians, such as tiger salamanders (*Ambystoma tigrinum*), as well as insects such as predaceous diving bugs (Belamostidae), predaceous diving beetles (Dytiscidae) and dragonflies (Odonata) (Didiuk 2003b).

5.3 Habitat Requirements

5.3.1 General

Henrich (1968) described the Canadian toad as a water-adapted species of prairie ponds and lakes that was less terrestrial than other bufonids. Underhill (1961) described Canadian toads as aquatic species always found close to water. In Alberta, Canadian toads inhabit seasonal wetlands, permanent ponds, borrow pits, ditches, lake margins, dugouts, oxbows and slow-moving streams (Robert and Lewin 1979, Cottonwood Consultants 1986, Russell and Bauer 2000).

5.3.2 Hibernacula

Canadian toads over-winter in upland areas with sandy soils rather than in the characteristic wet, muddy substrates of permanent wetlands (Breckenridge and Tester 1961, Tester and Breckenridge 1964). Kuyt (1991) observed one overwintering site near Fort Smith along the Northwest Territories-Alberta border in a sandy hillside along a road approximately several hundred meters from the nearest permanent waterbody. As previously discussed, in Minnesota, Canadian toads hibernate almost entirely in so-called 'Mima-type mounds' (Tester and Breckenridge 1964). These mounds of loose soil are generally no more than 60 cm high and are approximately 3 m to 15 m in diameter. Although thought to be partly a result of northern pocket gopher burrowing activities (Breckenridge and Tester 1961), these mounds may be primarily the result of soil displacement caused by Canadian toads burrowing beneath the soil annually for hundreds or thousands of years (Tester and Breckenridge 1964). The loose soil structure in these mounds is thought to facilitate burrowing (Tester and Breckenridge 1964). Some mounds appear to be preferred over others, with preferred mound characteristics unknown but possibly related to

mound profile or vegetation type (Tester and Breckenridge 1964). Although these mounds appeared to be the preferred overwintering sites, Canadian toads were also observed overwintering in in wind-blown soil deposits along fencelines (Tester and Breckenridge 1964).

Kuyt (1991) observed more than 500 Canadian toads at one overwintering site in northeastern Alberta, suggesting that toads congregate at communal hibernacula. Communal overwintering sites have also been documented in Minnesota (Breckenridge and Tester 1961). The tendency of Canadian toads to overwinter in communal sites may reflect a shortage of suitable substrates for burrowing in some areas (Hamilton *et al*. 1998). Suitable hibernacula sites may therefore be a limiting factor in the survival of Canadian toads at northern latitudes. Adult Canadian toads appear to show strong fidelity to traditional overwintering sites (Kelleher and Tester 1969).

5.3.3 Breeding Habitat

Throughout their range, Canadian toads are dependent on aquatic habitats for breeding (Hamilton *et al.* 1998). Shallow water in seasonal wetlands or along the margins of lakes and permanent wetlands are preferred for breeding (Didiuk 2003b). In northeastern Alberta, Canadian toads were found spawning in natural ponds, borrow pits, lake margins and streams (Roberts and Lewin 1979). According to GoA (2013a), Canadian toads can be found in shallow lakes, ponds and ephemeral wetlands. Lauzon and Balagus (1998) often found Canadian toads in ephemeral wetlands along with plains spadefoots in the Youngstown, Alberta area. In Minnesota, Canadian toads appeared to prefer ponds with stable water levels, gradually emerging shores with mud flats, common cattail and bulrushes in the emergent zone, and floating mats of sago pondweed (*Stuckenia pectinata*) and filamentous algae in the open water zone as opposed to ponds with shallow marginal zones, dense stands of sedges (*Carex* spp.) and an absence of mud flats (Breckenridge and Tester 1961). In Montana, the single recorded observation of a Canadian toad was made along the muddy shoreline of a 0.8 ha (2 ac) pond ringed by sedges and bulrushes and surrounded by rolling native prairie (Black and Bragg 1968).

5.3.4 Foraging Habitat

After the breeding season, Canadian toads may move to adjacent upland areas or remain near their breeding waterbodies. Native grassland is preferred to cropland for foraging (Didiuk 2003b). In northeastern Alberta, Canadian toads were observed most frequently in waterbodies as well as grassy meadows and willow bogs adjacent to water in July and August (Roberts and Lewin 1979). In Minnesota, adult Canadian toads were most frequently observed along the margins of waterbodies following breeding (Breckenridge and Tester 1961). Canadian toads in Minnesota appear to prefer wetlands with dense vegetation during their active period (Tester *et al.* 1965). In contrast to adults, shortly after metamorphosing many young-of-the-year frequently disperse into upland areas adjacent to waterbodies (Breckenridge and Tester 1961). Other young-of-the-year prefer to remain along the margins of waterbodies where the eggs were laid (Breckenridge and Tester 1961). In northeastern Alberta, Constible *et al.* (2009) found that Canadian toads made substantial use of upland forest habitat in the post-breeding period (*i.e.*, ~30% of their time), although use of this habitat type was actually less than what would be expected based upon availability. Wetlands were the preferred habitat, with use higher than what would be expected based upon availability (Constible *et al.* 2009).

In a study of habitat use of the closely related Wyoming toad (*Bufo baxteri*), Parker and Anderson (2003) found that soil surface temperature and distance to the shoreline of a breeding lake were the best predictors of habitat use by that toad species in Wyoming. Relative probability of use increased with closeness to the shoreline and at soil surface temperatures of 14°C to 15°C. Toads in this study were dependent on water-saturated substrates for the entirety of their terrestrial life and preferred areas with higher vegetation cover (average = 52.6% for adults, 39.2% for juveniles).

5.3.5 Area Requirements

As previously discussed, Kuyt (1991) observed Canadian toads overwintering in hibernacula located several hundred meters from the nearest permanent waterbody. Constible *et al.* (2009) observed overwintering sites 654 m to 1,386 m away from breeding sites in northeastern Alberta. Both of these studies suggest that Canadian toads travel considerable distance to obtain suitable overwintering habitat. In contrast, Breckenridge and Tester (1964) found that Canadian toads in Minnesota hibernated in uplands sites located approximately 23 m to 35 m away from breeding ponds.

Roberts and Lewin (1979) observed Canadian toads up to 100 m from waterbodies, but abundance decreased at distances greater than 40 m. Although Canadian toads appeared to disperse up to 61 m (200 ft) from breeding ponds, Breckenridge and Tester (1961) observed approximately 80% of all their toads in Minnesota within 7.6 m (25 ft) of waterbodies. Similarly, Parker and Anderson (2003) found all of their Wyoming toads within 10 m of the breeding lake. Canadian toads can be found up to 1 km away from breeding wetlands in Saskatchewan (Didiuk 2003b).

In a tracking study of Canadian toads by Breckenridge and Tester (1961), 84% of daily movements were 30.5 m (100 ft) or less and only 4% of movements were 61 m (200 ft) or more. These authors calculated that home ranges averaged 172 m (565 ft) in diameter (range: 15 m to 390 m) based on the distance between the most widely-spaced tracking points. Tarnsitt (1962) observed that a population of Canadian toads along the southern shore of Lake Manitoba made daily movements of 76 m (250 ft) to 122 m (400 ft) between preferred nocturnal and diurnal habitats. Constible *et al.* (2009) found that while Canadian toads in northeastern Alberta generally move less than 50 m at a time, long distance, directional movements of greater than 100 m are sometimes undertaken. Parker and Anderson (2003) found that Wyoming toads occupied an area of 30 m x 500 m (1.5 ha) in summer, and individual toads moved 0 m/day to 151.8 m/day.

6 GRAZING AND HERPETILE GROUP 2

Livestock grazing is the dominant land use in southeastern Alberta where plains hognose snakes, plains spadefoots, Great Plains toads and Canadian toads occur. The direct and indirect effects of grazing on these species have not been well studied in Alberta or in their greater North American range.

Additional research is also needed to characterize the habitat requirements and biology of these species in Alberta to better understand the potential impacts of grazing. Since ranching allows for the retention of intact native prairie areas and critical wetlands, and as most native prairie fauna co-existed in their historical range with bison, livestock grazing is generally considered a beneficial land use for maintaining habitat for a diversity of wildlife species. However, the benefits of livestock grazing depend on appropriateness of stocking rates, timing of grazing use and distribution of cattle in relation to critical habitats.

Directly, intensive livestock use near wetlands can have a negative impact on the survival of amphibian larvae due to trampling or increased water turbidity and creation of anoxic conditions from nutrient loading, erosion and defecation (Didiuk 1999a, Didiuk pers. comm.). Trampling mortality, soil compaction or the removal of thermal or escape cover are other potential negative effects from livestock grazing on herpetofauna. The creation of dugouts in ephemeral wetland basins can also have negative effects on Great Plains toad and plains spadefoot breeding sites (James 1998, Didiuk pers. comm.). Dugouts created within wetland basins drain water from potentially suitable toad breeding sites (Didiuk pers. comm.). High-sided, deep dugouts with no emergent vegetation cover do not provide suitable breeding habitats for plains spadefoot or Great Plains toads (James 1998, Didiuk pers. comm.). Great Plains toads call from tall emergent vegetation (grasses or sedges) and both plains spadefoot and Great Plains toads require emergent vegetation and shallow, warm water to provide suitable sites for egg mass attachment and rapid egg and larval development (James 1998, Lauzon 1999). Submerged vegetation also provides important shelter for developing tadpoles.

Positive effects may occur from livestock use if trampling or grazing creates basking sites or breaks up dense sod creating suitable burrowing sites for snakes and toads (Denton and Beebee 1996). Livestock “wallowing” can also create depressions which form into suitable breeding sites (ephemeral wetlands) for toads following rainfall (Bragg 1940). Bragg (1940) noted that Great Plains toads in Oklahoma bred primarily in rain-formed pools in “buffalo wallows” in uncultivated areas.

Indirectly, livestock herbivory affects plant species composition and vegetation structure, which may influence the community composition or abundance of small mammal or insect prey. A diverse small mammal population not only provides a prey source for plains hognose snakes, but also increases the availability of burrow shelters for snakes and toads (Breckenridge and Tester 1961, Wright and Didiuk 1998, Didiuk 1999a). Uniform, heavy grazing for prolonged periods has been found to depress populations of most small mammals (Fagerstone and Ramey 1996). Light to moderate grazing intensity, in contrast, stimulates diversity in the vegetation canopy and consequently creates habitat for a greater variety of small mammals. Light to moderate grazing promotes diverse vegetation structure by creating a gradient of heavy, moderate to lighter use patches across the landscape.

The following discussion summarizes the potential interactions of grazing on plains hognose snakes, plains spadefoots, Great Plains toads and Canadian toads based on available studies and discussions with species experts.

6.1 Plains Hognose Snake Response to Grazing

According to information from captures at C.F.B. Suffield, plains hognose snakes were found in highest densities in open, sandy plains, where grazing pressure was described as "light", resulting in a "heavy cover of mixed grasses;" and secondly in "open plains with light to moderate grazing pressure near wetlands" (Wright and Didiuk 1998, p.2). Lesser densities of snakes were found in the following habitats: sandhill country; short or mixedgrass plains with a heavy sod layer and "moderate to heavy" grazing pressure and gravelly, glacial till-plains; and riparian areas (Wright and Didiuk 1998, p.2). These results suggest that appropriate management of stocking rates (*i.e.*, light to moderate grazing) may be important for maintaining suitable plains hognose snake habitat. Managing the impact of grazing near seasonal or permanent wetlands is also considered important (Didiuk 1999a). Seasonal wetlands with clear, clean water provide critical breeding sites for Great Plains toads, an important plains hognose snake prey item (Wright and Didiuk 1998).

Plains hognose snakes at C.F.B. Suffield were also found most frequently in association with northern pocket gophers. Pocket gophers are generally attracted to rangeland in good to excellent condition with vigorous plants with large root systems (Fagerstone and Ramey 1996).

6.2 Plains Spadefoot Response to Grazing

Plains spadefoots have been observed to successfully reproduce in ponds that were heavily disturbed by cattle in the Milk River area (Klassen 1998). However, breeding success has not been compared between disturbed and undisturbed breeding sites. Soil compaction by cattle in the vicinity of breeding sites may reduce the availability of suitable burrowing sites (Didiuk pers. comm.). Heavy use near breeding sites can also reduce vegetation cover, diminishing predator escape cover or foraging opportunities for young toads (Didiuk pers. comm.).

6.3 Great Plains Toad Response to Grazing

As water clarity is considered an important characteristic of Great Plains toad breeding sites, heavy cattle trampling and use near breeding ponds may be detrimental to the reproductive success of this species (Bragg and Smith 1943, Wershler and Smith 1992). Heavy trampling, erosion and cattle defecation near breeding sites has the potential to reduce water quality and vegetation cover at these sites. Deep hoof prints and hummocking along wetland edges may also entrap or make it difficult for toads to access or exit ponds (James 1998). Yet, Bragg (1940) noted that Great Plains toad tadpoles have fully developed in waters that were disturbed by livestock after egg laying and hatching. Newly transformed young may face trampling risks from heavy use near breeding ponds as young toads often remain near pools until they evaporate (Smith and Bragg 1949).

6.4 Canadian Toad Response to Grazing

As previously discussed, Canadian toads are dependent on shallow lakes, ponds and ephemeral wetlands for breeding, although they may make use of other riparian/aquatic habitats such as borrow pits and dugouts as well (Roberts and Lewin 1979, GoA 2013a). Therefore, heavy grazing in fields containing breeding wetlands, especially during the spring when soils are

saturated and prone to compaction and erosion, is likely to negatively impact Canadian toads. As discussed above for Great Plains toads, pugging in soft soils along wetland margins may also entrap or make it difficult for Canadian toads to enter or exit breeding ponds. These pugs may also trap tadpoles or larvae, resulting in mortality as the pugs eventually dry up (Sarrell 2004). Moreover, because Canadian toads burrow beneath sandy soils in upland sites away from breeding wetlands (Breckenridge and Tester 1961, Tester and Breckenridge 1964), heavy stocking rates in fields containing hibernacula sites has the potential to cause compaction and reduced suitability of these sites for overwintering. Heavy grazing during the breeding season or when Canadian toads are dispersing to their overwintering sites also has the potential for trampling of toads.

7 GRAZING SYSTEMS AND HERPETILE GROUP 2 HABITAT MANAGEMENT

Table III-14 provides an overview of six grazing systems and their potential implications for maintaining or enhancing habitat for plains hognose snakes, plains spadefoot, Great Plains toads and Canadian toads. A grazing system is a tool used to control the timing, intensity and frequency of livestock grazing (Holechek *et al.* 2003).

Table III-14 Grazing Systems and Herpetile Group 2 Habitat Management

Grazing System	Discussion
Continuous (Season-Long) Grazing	
Advantages:	As livestock densities are typically lower in continuous grazing systems there is less risk of soil compaction (Abouguendia and Dill 1993). Soil compaction impedes the ability of plains hognose snakes and toads to burrow into the ground for shelter in areas of sandy soil overlain by finer textured soils. Under light to moderate stocking rates, continuous grazing often results in heterogeneous vegetation structure due to patchy grazing. Patches of unused or lightly grazed, taller vegetation provide shade and predator escape shelter for snakes and toads. More heavily grazed sites provide basking sites and may break up dense sod and create burrowing sites (Denton and Beebee 1996). More heavily used areas would also maintain active sand dunes or areas with bare sand, perhaps increasing suitable burrowing sites for toads and snakes. Structural diversity also favours a greater number of rodent species, potentially providing a more stable and varied prey base for snakes (Fagerstone and Ramey 1996). Because continuous grazing has the potential to negatively impact wetlands during the spring season when soils are saturated, the benefits of continuous grazing depend on maintaining sustainable stocking rates and implementing livestock distribution tactics (*i.e.*, salt, herding or water developments) to distribute cattle away from critical toad breeding sites, particularly during the breeding season.

Disadvantages:	Continuous grazing can be particularly damaging to riparian areas (including wetlands, streams and rivers), as livestock will congregate and linger near water throughout the duration of the grazing season from spring to fall (Ohmart 1996, Fitch and Adams 1998, Holechek *et al.* 2003). For this reason, continuous grazing nearly always results in overuse of riparian areas (Ohmart 1996). Heavy, persistent grazing in riparian areas can result in soil compaction, trampling and removal of vegetative cover, increased erosion and degraded water quality. These factors have the potential to negatively affect habitat quality and reproductive success of plains spadefoot, Great Plains toads and Canadian toads. At heavy stocking rates, continuous use of fragile sandy soil ecological sites can lead to accelerated erosion of these rangelands, potentially diminishing the long-term productivity of these sites for vegetation growth and sustainable livestock production (Houston pers. comm.). Plains hognose snakes have been found in lesser densities in moderate to heavily grazed dry mixedgrass prairie in southern Alberta (Wright and Didiuk 1998).
Rotational Grazing - Deferred Rotation	
Advantages:	Deferred spring grazing has the advantage of preventing cattle use near wetlands used for breeding by Great Plains, plains spadefoot and Canadian toads during the breeding season (*i.e.*, late April to mid- July). Stone (1991) recommended summer exclusion of livestock from fields containing Wyoming toads, a species closely related to the Canadian toad. This minimizes possible negative effects due to trampling of egg masses or newly developed toads and reduced water quality from erosion or defecation (Didiuk pers. comm.). A study conducted in southern Alberta found that a deferred-grazed native grassland field associated with a wetland creation project supported a greater number of herpetiles (amphibians and reptiles) than tame pasture or continuously-grazed native grassland fields (Fisher and Roberts 1994). The deferred-grazed native grassland field supported various species including boreal chorus frogs, tiger salamanders, plains garter snakes and plains spadefoot toads (Fisher and Roberts 1994).
Complementary Grazing	
Advantages:	Using tame pasture in the spring allows for deferred use of native prairie during the critical breeding season (see above). Herbicide and pesticide should not be applied to tame pastures near breeding sites or foraging habitats used by toads and snakes. To retain adequate cover and minimize disturbance to the hydrology of breeding wetland sites, a minimum 15 m unseeded buffer should be retained around wetlands contained within tame pastures (Kingsbury and Gibson 2002).

Disadvantages:	Complementary grazing requires the use of tame pasture, which may not be available in all livestock operations. Tame pasture is often not suitable in many places in the mixedgrass and dry mixedgrass prairie due to the fragility of the soils and dry conditions. Tame pastures are considered lesser quality habitat for plains hognose snakes, plains spadefoots, Great Plains toads and Canadian toads (Cottonwood Consultants 1986). Fisher and Roberts (1994) reported that tame pastures that formed part of the Medicine Wheel Project in southern Alberta supported significantly fewer herpetiles than deferred-grazed native pastures. Pasture rejuvenations (*i.e.*, re-seeding) can be expensive and may result in the cultivation of toad breeding habitat, including temporary/seasonal wetlands as well as the margins of semi-permanent/permanent wetlands. If implementation of a complementary grazing system requires the conversion of native prairie to tame pasture it is not likely to provide added benefit to Alberta herpetiles.
Rotational Grazing	
Advantages:	Kingsbury and Gibson (2002) recommend light to moderate rotational grazing, with no more than one third of available habitat grazed in one year, as an effective means to manage habitat for a diverse population of amphibians and reptiles. Rotational grazing is often recommended as a means to improve the health of riparian areas as it promotes periods of rest and recovery. Rotational grazing may therefore improve the quality of toad breeding sites. This type of grazing system also offers a means to control the timing of grazing in critical habitats to avoid sensitive periods. This includes avoiding use near plains hognose snake hibernacula in the spring and fall and avoiding use near toad breeding areas in the spring and summer (late April to mid-July). Periods of rest and recovery are also beneficial for maintaining the long-term productivity and stability of sandy soil range sites, while allowing for periodic disturbance to maintain some areas of active sand dune sites.
Disadvantages:	Under heavy stocking rates, rotational grazing can lead to uniform grazing effects, diminishing vegetation structural heterogeneity. A uniform reduction in vegetation cover diminishes available thermal cover and shelter sites for snakes and toads and can reduce the suitability of an area for populations of some small mammals such as voles and pocket gophers (Fagerstone and Ramey 1996). Rotational grazing that results in higher livestock densities in a given field could lead to greater trampling danger to toads if grazing occurs during the breeding season or during dispersal to hibernacula.

Rotational Grazing – High Intensity Grazing	
Advantages:	Intensive, short periods of grazing in the fall may be beneficial for reducing encroachment of tall exotic grasses such as smooth brome (*Bromus inermis* ssp. *inermis*) or timothy (*Phleum pratense*) in plains hognose snake or toad foraging areas or dispersal corridors. Intensive grazing in the spring may have a similar effect but may jeopardize the health and productivity of breeding wetlands.
Disadvantages:	Sandy soil ecological sites may be subject to greater risks of erosion and destabilization of vegetation when managed under intensive grazing systems (Houston pers. comm.). Intensive grazing also has the potential to increase soil compaction and create uniform vegetation structure across the range, diminishing burrowing opportunities for plains hognose snakes and toads and reducing thermal and escape cover. Intensive grazing systems can also result in degradation of riparian areas, possibly diminishing the quality of toad breeding sites. Intensive grazing may also lead to increased trampling risks to egg masses, tadpoles and newly transformed young due to higher livestock densities. The impacts of intensive grazing depend on the timing of grazing use in pastures that contain breeding sites, and on the period of rest following grazing.
Riparian Pasture Grazing	
Advantages:	Grazing systems designed to improve the health of wetland/riparian areas may benefit plains spadefoots, Great Plains toad and Canadian toads by: i) reducing grazing impacts to breeding sites during the breeding period; and /or ii) improving water quality and vegetation cover at and around breeding sites. For example, restricting grazing of pastures with breeding wetlands until after breeding and metamorphoses has been completed may benefit these toad species and lead to improved riparian health.
Disadvantages:	Given the ephemeral nature of wetlands used as breeding sites by plains spadefoot, Great Plains toads and Canadian toads, it may be difficult to develop grazing management plans for these types of seasonally variable riparian areas.

8 BENEFICIAL MANAGEMENT PRACTICE RECOMMENDATIONS

Sandhill and sand plain areas with ephemeral wetland basins in the Milk River and South Saskatchewan River watersheds are considered prime habitat for plains hognose snakes, plains spadefoots and Great Plains toads (Cottonwood Consultants 1986). Canadian toads will also make use of ephemeral wetlands, although they make use of seasonal and permanent wetlands as well (Lauzon and Balagus 1998, Didiuk 2003b, GoA 2013a). Conservation and proper management of native prairie habitat and wetlands in these areas is, therefore, important to the long term survival of these species. Sandhill and sand plain habitats are often considered fragile

ecosystems that are especially susceptible to damage from overuse by livestock, human activities or industrial development (Cottonwood Consultants 1986).

The following general land-use and grazing recommendations offer a variety of means to protect or maintain plains hognose snake, plains spadefoot, Great Plains toad and Canadian toad populations and their habitat within the Milk River and South Saskatchewan River watersheds. These recommendations apply to the full-extent of their range within the Grassland Natural Region of Alberta. Further research (see Section 9) is required to gain a better understanding of the ecology, habitat use and population dynamics of these four herpetile species, as well as to better understand the influence grazing has on these species, their habitats and their primary prey.

8.1 General Recommendations

Native Prairie Conservation

- Protect remaining mixedgrass and dry mixedgrass native prairie and sand plain and sandhill areas in the Milk River and South Saskatchewan River watersheds from cultivation or development. Plains hognose snakes, plains spadefoot, Great Plains toads and Canadian toads are all primarily associated with native prairie habitats. These species either do not occur or are found less frequently in cultivated areas (Cottonwood Consultants 1986). Although populations of plains spadefoots and Great Plains toads have been found in cultivated areas in southern Alberta (Downey 2006), it is not known if these habitats are sink habitats and if the two species will persist indefinitely. Cultivation is thought to render an area largely or entirely unsuitable for plains hognose snakes, plains spadefoot, Great Plains toads and Canadian toads due to removal of cover, destruction of wetlands, soil compaction and associated risks from machinery and chemical use (Wright and Didiuk 1998, Russel and Bauer 2000). Moreover, cultivation also increases sedimentation in wetlands and decreases the hydroperiod (*i.e.*, the period of time a wetland holds water), which causes smaller body sizes and altered immune systems in plains spadefoots and Great Plains toads and, in turn, likely reduces reproductive success and survivability of these two species (Luo *et al.* 1997, Gray and Smith 2005, McMurry *et al.* 2009). Cultivation can also reduce or locally eliminate colonial small mammal populations and their burrows, which are an important source of shelter and/or food for plains hognose snakes, plains spadefoots and Great Plains toads. Tall and dense cropland or tame pasture vegetation may also render areas unsuitable to snakes and amphibians by impeding their movement and dispersal across the landscape.
- Maintain or re-establish natural corridors between suitable native prairie habitat patches (Kingsbury and Gibson 2002). Retain uncultivated areas such as ditches or rights-of-way adjacent to existing cultivated areas to create dispersal corridors for plains hognose snakes and support rodent prey (Wright and Didiuk 1998).

Wetland Conservation and Restoration

- Restore formerly drained wetlands in plains hognose snake, plains spadefoot, Great Plains toad and Canadian toad ranges and prevent further wetlands from being drained or cultivated. Ephemeral wetlands cultivated during dry years have less ability to hold water

in wet years (Taylor and Downey 2003). Ephemeral or seasonal wetlands provide critical breeding habitat for plains spadefoot and Great Plains toads. Great Plains toads are known to congregate in large masses and use the same breeding sites in successive years (Krupa 1994).

- Avoid converting ephemeral breeding wetlands into permanent waterbodies (Lauzon 1999). Permanent waterbodies may harbour a greater abundance of aquatic and terrestrial predators than seasonal wetlands and are considered lower quality breeding sites for plains spadefoot and Great Plains toads (Woodward 1983). Altering wetlands may also affect water quality, vegetation composition, gradient and other factors that may influence their suitability as toad breeding sites.
- Maintain natural water levels and fluctuations (*i.e.*, avoid altering the natural hydrology of toad breeding sites) (Kingsbury and Gibson 2002).
- Identify and catalogue or map ephemeral wetlands that provide potential breeding habitat for plains spadefoot, Great Plains toads and Canadian toads.
- Protect a buffer of native vegetation around identified wetlands. Kingsbury and Gibson (2002) recommend leaving a minimum buffer of 50 feet (15 m). Maintaining a vegetated buffer around wetlands alleviates erosion and improves water quality by filtering out sediment and contaminant loads from runoff (Kingsbury and Gibson 2002). Vegetation cover around wetlands also provides cover and foraging habitat for plains hognose snakes and toads (Wright and Didiuk 1998).
- Maintain or re-establish natural corridors or native prairie patches between ephemeral wetlands used for breeding (Kingsbury and Gibson 2002). Isolated wetlands or wetlands fragmented by unsuitable habitat can isolate amphibian populations and cause small populations to die out.
- Avoid the introduction of game fish into natural wetlands used as toad breeding ponds (Kingsbury and Gibson 2002). Fish are efficient predators of amphibian eggs and larvae and can have a severe impact on amphibian reproductive success (Kingsbury and Gibson 2002).
- Where feasible, retain natural wetlands within new urban and residential developments.

Pest Control

- Avoid the use of pesticides or herbicides in marginal agricultural areas near plains hognose snake, plains spadefoot, Great Plains toad and Canadian toad habitats (James 1998, Wright and Didiuk 1998, Lauzon 1999). Chemical and sediment runoff from agricultural areas into wetland basins used for breeding degrades water quality and may adversely affect the breeding success or health of amphibians and the species that prey on them (*i.e.*, plains hognose snakes) (Kingsbury and Gibson 2002). Agricultural pesticides and herbicides have been found to have adverse effects on various amphibian species in Canada, including resulting in mutation, death or paralysis of tadpoles (Berill *et al.* 1997, Bonin *et al.* 1997). Amphibians are especially susceptible to harmful pollutants as they breathe through their skin. Pesticide use may also reduce the availability of insect or rodent prey for toads and plains hognose snakes (Anderson *et al.* 1999).
- Promote effective organic farming practices. Organic farming practices promote natural alternatives to synthetic pesticides or fertilizers. Organic farming thereby discourages the

use of harsh chemicals that could poison or diminish food supplies for plains hognose snakes, plains spadefoot, Great Plains toads and Canadian toads.

- Discourage the use of poisoning programs to control high densities of Richardson's ground squirrels or northern pocket gophers. Ground squirrels and pocket gophers are considered potentially important prey items for plains hognose snakes (Wright and Didiuk 1998), and their burrows may provide refugia and potential hibernacula for plains hognose snakes, plains spadefoot and Great Plains toads (Wright and Didiuk 1998, Didiuk 1999a).
- Use biological control agents as opposed to insecticides to control mosquitoes in urban wetlands if toad species of concern are known to be present (Stone 1991).
- Promote snakes and other natural predators as biological pest control agents.

Habitat Protection and Industrial Development Mitigation

- Abide by set-back distances and timing restrictions recommended by Alberta Environment and Parks (AEP) for human activities, including industrial developments, near to plains hognose snake hibernacula and rookeries (GoA 2011). For hibernacula, AEP recommends a 200 m year-round set-back for low and medium disturbances and a 500 m year-round set-back for high disturbance levels. For rookeries, a set-back of 200 m is recommended for all levels of disturbance from March 15th to October 31st. From November 1st to March 31st, rookeries have a set-back of 50 m for low and medium disturbances and a 200 m set-back for activities with high disturbance levels.
- Abide by AEP set-back distances for Great Plains toads and plains spadefoots (GoA 2011). AEP recommends a 100 m year-round set-back from Class III wetlands (Seasonal Ponds and Lakes) (Stewart and Kantrud 1971) in native prairie for all levels of disturbance. (Class III wetlands are referred to as Seasonal – Marshes under the Alberta Wetland Classification System; Alberta Environment and Sustainable Resource Development [ESRD] 2015). Aside from direct destruction of breeding ponds, oil and gas development may also affect toads due to disruption of groundwater resources, ground and surface water contamination and the consumptive use of water for drilling (Wershler and Smith 1992).
- Consider following the set-back guidelines for Canadian toad breeding ponds suggested for the Boreal Forest and Foothills Natural Regions (NRC 2006), which have a recommended year-round set-back distance of 100 m for all levels of disturbance (GoA 2013b).
- Where possible, avoid construction during the sensitive spring breeding season (late April to mid-June) to reduce possible impacts to plains hognose snakes, toads and other wildlife species (Lauzon 1999).
- Use Protective Notations under the *Public Lands Act* to protect key reptile and amphibian habitat.
- In areas where plains hognose snakes, plains spadefoots, Great Plains toads and Canadian toads are known to occur, routinely check and remove snakes and toads from pipeline trenches during construction (Wright and Didiuk 1998). Didiuk (1999) reported several instances of plains hognose snakes and, more commonly, Great Plains toads trapped at the bottom of underground "caissons" (structures that contain equipment and gas piping) at C.F.B. Suffield.

- Known hibernacula sites in Alberta are listed in the provincial Fish and Wildlife Management Information System (FWMIS) database. Developers and industry and their partners should conduct queries of this database as part of their environmental planning for proposed developments.
- Conduct pre-development wildlife surveys for plains hognose snakes, plains spadefoot, Great Plains toads and Canadian toads where suitable habitat for these species exists.
- The Milk River Watershed HSI maps that were developed for plains spadefoot and Great Plains toads should be used to inform environmental planning of oil and gas activities and other developments to ensure that areas with high potential for critical habitats are avoided (Taylor 2004a, b). Once additional information is available, HSI or RSF models should be developed for the plains hognose snake and Canadian toad.
- Ensure adherence to the Alberta Wetland Policy (GoA 2013c). Up to 70% of wetlands in Alberta's prairie pothole region have been converted to agricultural or other human-modified landscapes since European settlement (Locky 2011; GoA 2009; National Wetlands Working Group 1988; Schick 1972). Wetlands in Alberta are protected from disturbance under the Policy, which stresses a hierarchical approach to wetland disturbance, with avoidance as the preferred approach followed by minimization and then replacement. The Policy applies to all activities that have the potential to impact wetlands, whether carried out by a landowner or an energy company. The Policy applies to all natural wetlands in the province, including marshes and shallow open water wetlands, as well as all restored wetlands and wetlands constructed as part of wetland replacement initiatives.

Road Mortality

- A provincial road-kill monitoring program has been initiated in Alberta for all snake species to show problem areas for snake mortality and to provide distribution information for various species (Alberta Endangered Species Conservation Committee (ESCC) 2000). This program should be expanded to include amphibians.
- In areas with high densities of plains hognose snakes in proximity to roads, appropriate speed limits and warning signs should be posted to notify vehicles to use caution. Roads and associated mortality due to vehicle collisions are considered the second most limiting factor to plains hognose snakes in Alberta (Wright and Didiuk 1998).
- Road kills are considered a significant cause of mortality for adult and juvenile plains spadefoot and Great Plains toads in the central part of their range in the United States (Bragg 1940, Smith and Bragg 1949, James 1998). The impact of road mortality on these species in Alberta requires further study. Where possible, traffic should be minimized on roads near critical toad breeding areas in the spring (from May to June), particularly during or following periods of rain. Large numbers of Canadian toads may also be killed along roads, but road mortality is thought to have only local effects (Didiuk 2003).

Public Awareness

- Plains hognose snakes are occasionally killed as a result of being mistaken for the prairie rattlesnakes (Wright and Didiuk 1998). Public awareness campaigns are helpful for reducing this type of problem as well as promoting the plight of rattlesnakes, another

species listed as May Be At Risk in the Milk River and South Saskatchewan River watersheds (GoA 2016).

- Establish and maintain good relationships with landowners in southern Alberta whose land contains species at risk. Work with and support groups such as MULTISAR, Cows and Fish and others to increase awareness of the importance of permanent waterbodies to Canadian toads among landowners in the province. Permanent waterbodies are sometimes drained to provide additional cropland in agricultural areas. Permanent waterbodies provide important breeding habitat for Canadian toads (Breckenridge and Tester 1961, Roberts and Lewin 1979). Conservation of permanent wetlands will also result in other ecosystem benefits such as flood attenuation and control (Adamus and Stockwell 1983). Landowners should also be informed of their obligations under the Alberta Wetland Policy regarding wetland disturbance.
- Increase awareness in the agricultural community of the importance of wetlands in general and their significance to amphibians. Ephemeral, temporary and seasonal wetlands, as well as the margins of semi-permanent and permanent wetlands, are often cultivated when they are situated in crop fields. If landowners are able to better identify wetlands (including the drier margins) they will be better able to avoid them during agricultural activities. Although their significance is unknown (Didiuk 2003b), these habitats may be important for species of concern such as Canadian toads, plains spadefoot and Great Plains toads.
- Provide compensation to landowners who undertake habitat enhancement measures for wildlife species of concern on their land (Stone 1991). Provide public acknowledgement of landowners who work cooperatively with wildlife managers to conserve species at risk.
- Poster campaigns, distribution of pamphlets and other public information initiatives are useful for encouraging public support in monitoring and documenting new records of rare amphibians and reptiles in Alberta. Volunteer support is considered especially important to the long-term success and effectiveness of the *Alberta Amphibian Monitoring Program* and the *Bird Studies Canada Marsh Monitoring Program* (Takats and Priestley 2002, Priestley 2002). Both of these monitoring initiatives aim to provide long-term information about amphibian populations, distribution, and habitat use in Alberta.

Population Monitoring

- Better determine the population size and trend of Canadian toads throughout their range in Alberta. Numbers are thought to decreasing across parts of the species' range but there is a lack of sufficient data (Hamilton *et al.* 1998). Consider conducting formal surveys for Canadian toads in select areas based on the ESRD Sensitive Species Inventory guidelines (GoA 2013a). All survey data should be submitted to AEP for inclusion in FWMIS. Consider reporting 'zero data' when areas were searched for amphibians and reptiles but no animals were found (Browne 2009). Also consider including photographic records with data submissions (Browne 2009).

8.2 Grazing Recommendations

Livestock grazing systems that will benefit plains hognose snakes, plains spadefoot, Great Plains toads and Canadian toads should aim to:

- Minimize impact to critical breeding wetlands during the spring breeding season (*i.e.*, late April to mid-July).
- Maintain high riparian health of wetlands and high range health of adjacent native grassland.
- Provide a variable grassland structure to provide amphibians and reptiles and their prey with a diverse range of microsites, ranging from more open sites for basking to more shaded, moist sites with adequate cover.
- Retain adequate vegetation cover in and around critical breeding wetlands.
- Minimize soil compaction near overwintering, breeding and foraging sites.

Ultimately, the selection of an appropriate grazing system will depend on local ecological conditions, distribution of key habitat components and range management goals. Appropriate grazing systems should be developed site-specifically for participating ranches as part of the Multiple Species At Risk (MULTISAR) Conservation Program for the Milk River and South Saskatchewan River watersheds.

The following recommendations consider grazing management practices that will likely benefit plains hognose snakes, plains spadefoot, Great Plains toads and Canadian toads, and also maintain the condition of mixedgrass or dry mixedgrass prairie associated with sandy range sites.

- Use low to moderate stocking rates in pastures that contain or connect critical plains spadefoot, Great Plains toad, Canadian toad and plains hognose snake breeding, foraging and overwintering habitats (Wright and Didiuk 1998, Didiuk 1999a, Kingsbury and Gibson 2002). Adjust stocking rates during drought conditions.
- Maintain utilization rates of 25% to 50% for mixedgrass and dry mixedgrass prairie to ensure adequate carry-over of residual vegetation and litter (Adams *et al.* 2013a, b). Conservative stocking rates will help to prevent uniform grazing effects and retain sufficient upland cover for toads dispersing overland. Ecological sites with sandy soil textures (*e.g.*, Sands and Choppy Sandhills) are generally more drought prone than Loamy ecological sites and do not support the same stocking rates (Adams *et al.* 2013a, b). Retention of sufficient cover through the use of ecologically sustainable stocking rates offers shelter from predators and helps to prevent desiccation of toads. Vegetation cover is also considered important for northern pocket gophers, an important plains hognose snake prey.
- Use once-over grazing per season when implementing grazing systems in sandy ecological sites (Houston pers. comm.).
- Place salt or mineral sources at least 1 km from natural waterbodies, where possible (Adams *et al.* 1986). Placing salt away from water forces cattle to make better use of the range and reduces the amount of time cattle spend at critical breeding wetlands.

- Provide alternate watering sites (*e.g.*, dugouts or troughs) to reduce impact to Great Plains toad breeding wetlands. Cattle have been shown to prefer drinking from a water trough and will travel further to a trough rather than drink from a stream when given free access to both (Veira 2003). Off-site watering can provide a more reliable and cleaner water source for cattle. Improved water quality can improve livestock health and weight gain (Veira 2003).
- Avoid the creation of dugouts in critical toad ephemeral breeding ponds (James 1998, Didiuk pers. comm.). Dugouts are not considered suitable breeding habitats for Great Plains toads or plains spadefoots due to their depth, steep sides, lack of vegetation, nutrient loading and heavy use by cattle (James 1998, Didiuk pers. comm.). Didiuk (1999) noted that Great Plains toads at C.F.B. Suffield only bred in dugouts located in depressions with flooded areas immediately adjacent to the dugout.
- Modify existing dugouts to create a low shoreline gradient with shallow, marshy edges to improve habitat available to Great Plains toads (James 1998, Didiuk pers. comm.).
- Defer use near plains spadefoot, Great Plains toad and Canadian toad breeding ponds during the spring breeding season (late April to mid-July) to reduce impact to water quality, retain vegetation cover and prevent possible trampling mortality to spawning adults, egg masses, tadpoles or newly developed toads.
- Where necessary, fence out important plains spadefoot, Great Plains toad and Canadian toad breeding ponds to restore water quality or vegetation cover or to prevent damage during the breeding season.
- Use rest- or deferred once-over rotational grazing systems to improve native prairie and riparian area health as well as to defer livestock use during the breeding season (Fisher and Roberts 1994).

9 RESEARCH RECOMMENDATIONS

As the herpetofauna considered in this report are rare and have not been extensively studied in Alberta, more information is needed to characterize their habitat requirements, movement patterns and population dynamics in the Milk River and South Saskatchewan River watersheds using radio-telemetry and mark-recapture studies (Didiuk pers. comm.). This type of research will help in understanding how various land use activities such as livestock grazing influence these species. Developing a regular monitoring program that encompasses a variety of habitat types and moisture regimes is an important first step in building a better understanding of the population status, trends and distribution of plains hognose snakes, plains spadefoot and Great Plains toads.

The goal of grazing management research should be to assess exactly how much of an impact livestock grazing is having on critical amphibian and reptile habitat components (including hibernacula, breeding sites, burrowing sites, thermoregulation cover and foraging habitat), and to determine which grazing strategies (*i.e.*, timing, distribution, stocking rates) are most beneficial or are most effective at mitigating negative impacts.

Key research needs are described below for each species:

9.1 Plains Hognose Snake

- Develop a plains hognose snake monitoring program for the Milk River and South Saskatchewan River watersheds to assess fluctuations in snake populations and ranges.
- Conduct additional research to study the basic biology, ecology and habitat use by plains hognose snakes in Alberta. Research should focus on learning more about the preferred habitat and range of this species in Alberta and providing data on various undetermined aspects of this species' biology (Wright and Didiuk 1998).
- Specific research goals include: quantifying the home range of this species; characterizing foraging, breeding and egg laying habitats; mapping out hibernacula sites; and defining associations with key prey species or with rodents whose burrows provide important habitat.
- Develop HSI or RSF models for plains hognose snakes within the Milk River and South Saskatchewan River watersheds. HSI and RSF models provide valuable planning tools for land managers by mapping potential critical habitat of species of concern.
- Conduct applied research to understand how livestock grazing influences plains hognose snake foraging, thermoregulation, breeding and overwintering habitat quality and prey availability.
- To study the effects of grazing, research should be conducted in areas known to support healthy snake populations and which have representative areas of light, moderate and heavily grazed pastures in varying stages of range health (Wright and Didiuk 1998)

9.2 Plains Spadefoot, Great Plains Toad and Canadian Toad

- Conduct monitoring on a regular basis to assess population sizes and trends of plains spadefoot, Great Plains toad and Canadian toad populations in the Milk River and South Saskatchewan River watersheds. Monitoring efforts should encompass both drought and high precipitation years. Monitoring efforts should attempt to determine the number of breeding populations of toads in the Milk River Watershed, the response of these populations to drought and the ability of these populations to interact through immigration or emigration (James 1998). The GVI geodatabase, which maps out temporary, seasonal and semi-permanent/permanent wetlands, can be used to target toad surveys in southeastern Alberta.
- Identify, map and monitor important wetlands that are consistently used for breeding by large numbers of Great Plains toads, plains spadefoot and Canadian toads.
- Assess the proximity of overwintering sites to breeding sites for all three toad species.
- Assess seasonal habitat selection, movement patterns, home range size and overwintering requirements of all three toad species (Lauzon 1999).
- Determine the preferred vegetation cover characteristics in toad summer habitats (*i.e.*, percentage of bare ground versus graminoid, forb and shrub cover).
- Investigate the importance of rodent burrows as overwintering sites for plains spadefoots, Great Plains toads and Canadian toads (Breckenridge and Tester 1961, Didiuk 1999a, Didiuk pers. comm.).
- Determine the extent of communal overwintering by the Canadian toad in southern Alberta.

- Better determine what constitutes key habitat for Canadian toads in the Milk River and South Saskatchewan River watersheds (Browne 2009).
- Determine the extent of upland habitat use surrounding breeding ponds by juvenile and adult Canadian toads in southern Alberta.
- Determine minimum habitat patch size requirements and assess the effect of habitat fragmentation on toad movements and population dynamics.
- Investigate the effects of herbicide and pesticide contamination of breeding habitats on toad survival and reproductive success.
- Evaluate the effect of water management projects (*i.e.*, damming, dugouts, or creation of permanent wetlands in temporary wetland basins) on reproductive success and overwintering survival of toads.
- Evaluate the effect of water quality on Great Plains toad, plains spadefoot and Canadian toad reproductive success (Didiuk 2003a,b).
- Compare plains spadefoot, Great Plains toad and Canadian toad reproductive success and population dynamics in wetlands that receive heavy, moderate, light or no use by livestock during the breeding period.
- Investigate the effect of livestock disturbance and soil compaction on the availability of suitable burrowing sites for toads.
- Conduct field verification of the RSF probability of use maps created for the plains spadefoot and Great Plains toad (Skilnick and Dodd 2011, Skilnick *et al.* 2010).
- Study the effects of climate change and their anticipated effects on toads in the Grassland Natural Region (Didiuk 2003b).
- Determine winter mortality rates of adult and juvenile Canadian toads in Alberta, particularly in the Grassland Natural Region (Didiuk 2003b).

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C. NORTHERN LEOPARD FROG

1 INTRODUCTION

1.1 Background

The purpose of this report is to summarize the ecology and habitat requirements of the northern leopard frog (*Lithobates pipiens*) in southern Alberta. Based on this information and supporting scientific studies, various grazing systems are compared in terms of their potential implications to northern leopard frogs and their habitat. This discussion is followed by a summary of recommended beneficial management practices to protect northern leopard frogs and their habitat in the Milk River and South Saskatchewan River watersheds and throughout the Grassland Natural Region of Alberta (Natural Regions Committee [NRC] 2006). Lastly, a brief summary of additional information needs is presented.

The northern leopard frog is one of two species of *Lithobates* in Alberta, the other being the wood frog (*Lithobates sylvatica*) (Russell and Bauer 2000). Northern leopard frogs are larger than wood frogs and have a more restricted distribution in the province. Dorsal surfaces of northern leopard frogs are green- or brown-coloured with large, rounded dark spots outlined with light-coloured halos and two light-coloured dorsolateral ridges that stretch from the eyes to the lower back.

The northern leopard frog is listed as ‘At Risk’ under the General Status listing for Alberta (Government of Alberta [GoA] 2016) and is designated as a ‘non-game animal’ under Alberta’s *Wildlife Act*, making it illegal to kill, possess, buy or sell the species. The species has also been designated as ‘Threatened’ under the provincial *Wildlife Act*. Nationally, the northern leopard frog is designated as a species of ‘Special Concern’ by the Committee on the Status of Wildlife in Canada (COSEWIC 2002, 2009, 2016). Based on COSEWIC’s recommendation, the northern leopard frog was listed under Schedule 1 of the federal *Species at Risk Act* (*SARA*) in 2012 as a species of ‘Special Concern’. The species has been designated as a species of ‘Least Concern’ by the International Union for Conservation of Nature and Natural Resources (IUCN) (Hammerson *et al.* 2004).

Although formerly common and widely distributed in Alberta, significant declines in northern leopard frog populations occurred in the late 1970’s and early 1980’s (Roberts 1992, Alberta Sustainable Resource Development [SRD] 2003). Since this time, populations have remained at low levels in parts of southern Alberta and the species has been extirpated from much of western and central Alberta. Widespread population declines in Alberta during the 1970s coincided with similar massive declines in northern leopard frog populations in the upper Midwestern states and the other Prairie Provinces (SRD 2003). The current range of the northern leopard frog is primarily restricted to the southern and southeastern portions of the province. Populations of northern leopard frogs in the province currently appear to be stable (GoA 2014).

A number of population surveys for northern leopard frogs have been undertaken in southern Alberta. Formal northern leopard frog surveys have been conducted since 2000 and opportunistic surveys have been conducted since 2005 (GoA 2014). Surveys were conducted in the early

2000s as part of the Researching Amphibians in Alberta (RANA) project (Wilkinson and Hanus 2003, Wilkinson and Kempin 2004, Wilkinson and Berg 2004). During the RANA surveys, which were conducted for all amphibians in various locations throughout Alberta, northern leopard frogs were only observed in the Cypress Hills. However, the majority of monitoring sites were located outside of the historic range of the northern leopard frog.

During northern leopard frog-specific surveys in 2000 and 2001, the species was observed at 54 of 269 (20%) historic and recent sites (Kendell 2002a). Northern leopard frogs were found to be closely associated with major rivers, including the Oldman, the lower Red Deer, the Milk, the South Saskatchewan and the lower Bow. Other sites with northern leopard frogs included the Cypress Hills region and the extreme northeastern part of Alberta. Based on this survey it appears that northern leopard frogs have not re-colonized formerly occupied areas (Kendell 2002a). Moreover, it appears that further extirpation of northern leopard frogs has occurred in sites located south of Lethbridge, as well as south and west of Calgary. The occurrence data collected during this inventory did not allow for an accurate assessment of northern leopard frog population size (SRD 2003).

A province-wide survey for northern leopard frogs was undertaken in 2005 by the Alberta Conservation Association (ACA) and Alberta Sustainable Resource Development (ASRD) (now Alberta Environment and Parks [AEP]) in partnership with the Habitat Stewardship Program (Environment Canada) and the North American Waterfowl Management Plan (NAWMP) (Kendell *et al.* 2007). Surveyors re-visited 177 sites historically occupied by northern leopard frogs. Northern leopard frogs were observed at 73 of these sites and at three new sites. Largest concentrations of frogs were found in the Strathmore/Serviceberry Creek, Brooks, Medicine Hat/South Saskatchewan River valley, lower Red Deer River valley and Cypress Hills areas.

As part of the 2005 Alberta northern leopard frog inventory, a survey was completed in the Milk River Watershed in 2005 by the Multiple Species at Risk (MULTISAR) conservation program (Downey and Downey 2006). Of the 27 historic northern leopard frog sites surveyed in 2005, five sites had northern leopard frogs present, including one site with more than 100 individuals. One previously unrecorded northern leopard frog site was also identified. However, the majority of sites surveyed were considered unsuitable for northern leopard frogs due to a lack of water, recent cultivation or lack of sufficient depth to facilitate overwintering. The sites where northern leopard frogs persisted were along creeks and rivers and not wetlands.

Although the exact cause of northern leopard frog population declines is unclear, from their widespread nature and severity it would appear that these declines are not part of a regular cycle (SRD 2003). A combination of limiting factors, acting in unison or alone, may be responsible for the plight of the northern leopard frog. Proposed anthropogenic factors include: habitat loss or fragmentation; wetland drainage, eutrophication, acidification or pesticide contamination; water diversion, draw-downs or flooding (such as from dams or reservoirs); increased ultraviolet radiation due to reduction in the ozone layer; acid rain and pollution; livestock disturbance; road kills; harvest; and exotic species and game fish introduction (SRD 2003). Drought and disease are considered the primary natural limiting factors affecting northern leopard frog populations (SRD 2003).

In an effort to re-establish populations in historical habitats, a reintroduction project was initiated in 1999 by AEP and ACA. Over a five year period from 1999 to 2003, more than 12,000 captive-reared northern leopard frogs were released into suitable habitat within the historical range of the species in central and southern Alberta (Kendell 2002b, 2003, 2004). Of these 12,000 northern leopard frogs, 7,700 were released into the the upper headwaters of the Red Deer River near Caroline; 2,845 frogs were released along the North Saskatchewan River near Rocky Mountain House; and 1,310 frogs were released at a Ducks Unlimited Canada (DUC) property near the City of Red Deer. Monitoring of these three release sites confirmed the presence as well as breeding activity at the Caroline site. No confirmed observation or breeding records have been recorded for the Rocky Mountain House or DUC release sites (Kendell 2004). All three release sites occur outside of the Milk River and South Saskatchewan River watersheds.

A second northern leopard frog reintroduction project was initiated near the town of Magrath in 2003 (Romanchuk 2003, Romanchuk and Quinlan 2006). Egg masses were collected from three different sites in southern Alberta and reared and released primarily in the locally named Dudley's Pond, located along the floodplain of Pothole Creek on the edge of Magrath. A total of 8,502 northern leopard frog tadpoles were released over a three year period. Monitoring of this founder population confirmed overwintering and breeding success at the release site as well as dispersal to new sites upstream and downstream (Romanchuk and Quinlan 2006).

Additional reintroduction efforts continued until 2010 at which time the program shifted into the monitoring phase (GoA 2014). Additional reintroduction sites included Waterton Lakes National Park, Wyndham-Carseland Provincial Park and Beauvais Lake Provincial Park, among others (ACA 2010). Preliminary results showed that four of 10 reintroduction projects were successful, with signs of breeding and overwintering (GoA 2014). Based on the knowledge gained from northern leopard frog reintroduction projects in central and southern Alberta, including successes and failures, a formal reintroduction strategy was developed and released in 2007 (Kendell and Prescott 2007). This document outlines all of the considerations and protocols necessary to undertake a successful reintroduction project.

Because of its listing as 'Threatened' under the Alberta *Wildlife Act*, a provincial five-year recovery strategy for the northern leopard frog was developed and released in 2005, covering the years 2005 to 2010 (Alberta Northern Leopard Frog Recovery Team [ANLFRT] 2005). This plan identified all known threats and limiting factors, research needs, ongoing recovery efforts and a recovery action plan going forward. An updated recovery strategy covering the years 2010 to 2015 was released in 2012 (Alberta Environment and Sustainable Resource Development [ESRD] 2012). In addition, the federal government developed a management plan for the western boreal/prairie population of the northern leopard frog in accordance with its obligations under Section 65 of *SARA* (Environment Canada 2013).

1.2 Ecology

Northern leopard frogs overwinter in well-oxygenated waters (SRD 2003). Northern leopard frogs are not freeze-tolerant and do not truly hibernate, instead remaining inactive at the base of ponds and other waterbodies during the winter season (Churchill and Storey 1995). Frogs emerge from overwintering sites in the spring shortly after the ice begins to melt. Northern leopard frogs

are active from April to October (SRD 2003). Breeding is asynchronous and occurs from late April to late June during optimal temperatures (SRD 2003, Randall *et al.* 2014). Depending on environmental conditions, breeding may last only a few days or occur intermittently over several weeks. Mating typically begins when sexually mature males arrive at the breeding pond and start to call while floating on the surface (SRD 2003). Sexually mature females join them at the pond a few days to a few weeks later to lay their eggs. Eggs are generally attached to submerged vegetation and are laid in one mass containing between 600 and 7,000 eggs (Corn and Livo 1989). Females are thought to produce one clutch per year. Embryos show variable rates of development; eggs laid in warmer water and later in the season have faster development rates than those laid in cooler water and earlier in the season (Randall *et al.* 2014). Eggs laid in cooler water may take up to two weeks to develop (Randall *et al.* 2014). Hatching can occur five to nine days after egg laying, depending on water temperature (Hine *et al.* 1981). Flooding, drought or cold temperatures can result in egg mortality or loss of entire clutches. Tadpoles disperse from egg masses a few days after hatching. It takes 60 to 90 days for tadpoles to metamorphosize, with metamorphosis generally taking place in July and early August in Alberta (SRD 2003). Following breeding, adults usually remain near breeding ponds or disperse to summer feeding areas (Dole 1967, SRD 2003). Adults move to overwintering sites in late summer or early fall (SRD 2003).

Northern leopard frogs usually become sexually mature in their second year; however, reaching this stage depends more on body size than age (Corn and Livo 1989). Environmental conditions affect growth rate, which may vary from year to year. Northern leopard frogs may live up to four or five years of age in the wild (Russell and Bauer 2000). Annual adult mortality rates may be as high as 60% (Merrell and Rodell 1968). The ratio of young-of-the-year (YOY) to adult northern leopard frogs in Minnesota ranged from 15:1 to 20:1, with similar values reported for Wisconsin (Merrell 1977, Hine *et al.* 1981).

A population viability analysis conducted for northern leopard frogs in Alberta revealed that the species may not be able to exist in single, isolated populations for extended periods of time (Tischendorf 2007). The species vulnerability to local extirpations was attributable to density dependent regulations at the tadpole stage and to stochastic variations in demographic and environmental factors. Northern leopard frogs also have inherent growth potential which allows the species to quickly occupy and utilize existing habitat (Tischendorf 2007). Such a characteristic suggests that the species is adapted to exist in spatially distributed but connected populations (Tischendorf 2007). Tischendorf (2007) suggested that the minimum viable population size for northern leopard frogs is 30 adult female frogs (*i.e.*, two years or older) spread across two spatially distinct populations.

1.2.1 Diet

Northern leopard frogs shift from a primarily herbivorous diet of vegetation and algae as tadpoles to a carnivorous diet at metamorphosis (Fisher 1999). Adult frogs have a broad diet consisting primarily of invertebrates (Fisher 1999). Primary prey items include beetles (Coleoptera), leafhoppers (Homoptera), ants (Hymenoptera), true bugs (Hemiptera), grasshoppers (Orthoptera), moths and butterflies (Lepidoptera) and dragonflies (Odonata) (Moore and Strickland 1954). Adults have also been known to prey on worms (Oligochaeta,

Nematoda), snails (Gastropoda), small birds, garter snakes and fish and may occasionally cannibalize newly transformed young northern leopard frogs (Eddy 1976, Russell and Bauer 2000, SRD 2003). Northern leopard frogs may also vary their diets based on the season; for example, in Manitoba Eddy (1976) found that insects were found in 50% of northern leopard frog stomachs in early fall, whereas insects were found in 95.5% of stomachs in spring.

1.2.2 Predators

Vertebrate predators of northern leopard frogs include fish, great blue herons (*Ardea herodias*), belted kingfishers (*Ceryle alcyon*), shrews (*Sorex* spp.), weasels (*Mustela* spp.), red fox (*Vulpes vulpes*), and coyotes (*Canis latrans*) among others (Fisher 1999). Predators of northern leopard frog tadpoles include dragonfly nymphs, caddisfly larvae, beetles, leeches (Hirudinea), belted kingfishers, hooded mergansers (*Lophodytes cucullatus*), red-sided garter snakes (*Thamnophis sirtalis*), neotenic tiger salamanders (*Ambystoma tigrinum*) and introduced and native fish species (Dickerson 1907, McAllister *et al*. 1999, COSEWIC 2009).

1.3 Habitat Requirements

1.3.1 General

Northern leopard frogs need up to three distinct habitat types to fulfill their life cycle: a breeding waterbody in the spring for mating, deposition of eggs, and development of tadpoles; upland habitat near water for foraging during mid-summer; and a waterbody that does not freeze solid in so that they may overwinter (Kendell 2002c). In Alberta, northern leopard frogs are generally associated with clear water that is “relatively fresh to moderately saline” (SRD 2003). Springs appear to be especially important habitat variables for northern leopard frogs in Alberta, both for breeding and overwintering (see discussion below) (Kendall pers. comm.).

As part of the MULTISAR program, a Habitat Suitability Index (HSI) model was developed for the northern leopard frog habitat use in the Oldman River Watershed in southwestern Alberta (Pearson 2004). Variables in the HSI model included: natural subregion; waterbody type (*e.g.*, lakes, oxbows, irrigation canals, dugouts, *etc*.); distance to waterbodies greater than and less than 200 ha; waterbody size; and presence of fish. Preferred habitat was described in the following manner: habitat located in the Mixedgrass, Foothills Fescue, Foothills Parkland and, to a lesser extent, Montane Subregions; habitat consisting of lakes/oxbows, rivers, streams, canals, dugouts and reservoirs; habitat located less than 500 m from waterbodies 200 ha or less in size; habitat within 200 m of waterbodies 200 ha or more in size; all waterbodies, but with those greater than 200 ha in size ranked lower than those less than 200 ha in size; and habitat lacking fish which could act as predators of northern leopard frog larvae. The Oldman River Watershed contains more foothills habitat than either the Milk River Watershed or the South Saskatchewan Watershed; therefore, the HSI model developed for the Oldman River Watershed may not be applicable to the watershed discussed in this document.

A second HSI model was developed by AEP for northern leopard frog general habitat use in southern Alberta as a whole (Stevens *et al.* 2010). Eleven environmental variables were chosen and given equal weighting in the final model: distance to permanent hydrography features; density of hydrography features; permanent hydropolygon perimeter density; road density;

moisture index; mean annual temperature; percent graminoid cover; percent riparian cover; percent wetland cover; soil salinity; and presence of Solonetzic soils. Model validation showed that the model was good at predicting northern leopard frog habitat use. In general, northern leopard frogs are found in areas close to high densities of large, permanent wetlands and lakes; areas with low densities of roads; areas of high moisture and mean annual temperatures of 4°C to 5°C; high percent cover of graminoids; low percent cover of riparian and high percent cover of wetlands; sites with low soil salinity; and sites with low percentages of Solonetzic soils.

A Resource Selection Function (RSF) model was developed for northern leopard frog habitat selection in southern Alberta, also as part of the MULTISAR program (Gan *et al.* 2014). RSFs are thought to be more robust than HSI models due to their rigorous and empirical evaluation of model performance using statistics (Boyce *et al.* 2002). RSFs estimate relative probability of occurrence by species of concern on the landscape based on various physical and anthropogenic variables. The best-fit northern leopard frog RSF model, as determined by Gan *et al.* (2014), contained five variables: topographic wetness; percent grass cover; distance to cropland; distance to anthropogenic development; and distance to roads. According to the top model, relative probability of northern leopard frog occurrence increases with topographic wetness (*i.e.*, low-lying areas, which includes wetlands), percent grass cover, closeness to developments and roads and greater distances from cropland. It was not clear from the modelling process if northern leopard frogs were selecting for habitat along roads (*e.g.*, ditches) or if the apparent selection was a result of data collection techniques (*e.g.*, roadside surveys that record the geographic location of the road and not the wetland) (Gan *et al*. 2014).

Kolozsvary and Swihart (1999) modelled northern leopard frog habitat selection in Indiana in a landscape comprised of a mix of tall grass prairie, forest, wetlands and cropland. Their results showed that northern leopard frogs in their study area selected for isolated forest patches that were in close proximity to wetlands. Modelling of habitat use by northern leopard frogs in Colorado showed that the species there had a positive association with total area of grassland and wetlands and a negative association with urban/suburban development and introduced bullfrogs (*Lithobates catesbeianus*) and fishes (Johnson *et al.* 2011). In Maine, Blomquist and Hunter Jr. (2009) found that moisture was a dominant factor in habitat selection at all spatial scales (*i.e.*, home range, weekly activity centre and daily microhabitat). In Washington, Germaine and Hays (2009) developed site occupancy models at two different spatial scales: site (pond) and local (1 km^2). At the site level, northern leopard frogs occupied ponds that were slightly deeper, had less tall emergent vegetation, more herbaceous cover and fewer neighbouring ponds containing introduced, predatory fish. At the local scale, northern leopard frogs occupied ponds with greater midsummer depths, more herbaceous cover surrounding the ponds and areas with fewer ponds occupied by bullfrogs and carp (*Cyrprinus carpio*).

1.3.2 Overwintering

Northern leopard frogs overwinter in well-oxygenated water that does not freeze to the bottom, such as in springs, streams or in deeper lakes and ponds (SRD 2003). As northern leopard frogs absorb oxygen through their skin during hibernation underwater, they are susceptible to mortality due to anoxic water conditions or the complete freezing of waterbodies (Kendell 2002c). Spring-fed wetlands are considered critical overwintering habitat in southern Alberta, particularly during

periods of prolonged drought when deeper water bodies become scarce (Didiuk 1999, SRD 2003). Frogs overwinter under rocks, logs, leaf litter or vegetation, or in depressions in sand or mud (SRD 2003).

1.3.3 Breeding

Separate sites are typically used for overwintering and breeding, although occasionally, especially in spring-fed wetlands, the same waterbody may be used (Didiuk 1999a, SRD 2003). Northern leopard frogs usually breed in shallow and warm standing water such as in permanent and semi-permanent wetlands, marshes, sloughs, dugouts, springs, lake margins and shallow bays, beaver ponds, slow-flowing creeks, and the backwaters and oxbows of rivers (Cottonwood Consultants 1986, Kendell 2002c, SRD 2003). In the Suffield National Wildlife Area (SNWA), northern leopard frogs were restricted in their distribution to meander scar wetlands along the South Saskatchewan River floodplain and small drainages flowing into the river (Didiuk 1999). Male (and probably female) northern leopard frogs tend to return to the same ponds year after year for breeding (Waye and Cooper 2001).

According to Kendell (2002c, pers. comm.), ideal northern leopard frog breeding habitat has the following characteristics:

- Some degree of permanence (*i.e.*, it is unlikely to dry up before tadpoles metamorphose).
- Abundant aquatic and emergent vegetation.
- Shallow open water that receives direct morning and afternoon sunlight.
- Standing water that freezes solid during most winters or dries up every few years to prevent or reduce predatory fish establishment.
- Non-acidic water (*i.e.*, pH range between 6.5 and 8.5).
- Shallow water depths between 10 cm and 65 cm.
- A gradual sloping shoreline to support emergent and upland vegetation.
- Situated within 1.6 km of overwintering habitat.
- Spring-fed.

Similar to over-wintering ponds, springs are important habitat features often associated with northern leopard frog breeding ponds (Kendall pers. comm.). Indeed, some breeding ponds are made possible by the presence of spring alone (Kendall pers. comm.). Springs are especially important habitat features during drought conditions (Kendall pers. comm.). Spring sources for northern leopard frog breeding ponds can be either natural or anthropogenic (*e.g.*, irrigation pivots, leaking irrigation canals) in origin (Kendall pers. comm.)

Northern leopard frogs lay their egg masses in shallow water on vegetation and occasionally on the pond bottom (SRD 2003). Gilbert *et al.* (1994) found that northern leopard frog egg masses were deposited at depths less than 65 cm after the water temperature reached 8°C. Slow-flowing or stagnant waterbodies that contain water until late July or August and that are unsuitable for fish are considered favourable northern leopard frog spawning sites. Preferred spawning habitats often include areas dominated by herbaceous, non-emergent plants such as sedges (*Carex* spp.) and emergent plants such as cattails (*Typha* spp.) which provide points of attachment for egg masses (Hine *et al.* 1981, Gilbert *et al.* 1994).

1.3.4 Foraging (Non-breeding) Habitat

Northern leopard frog summer feeding sites are most often located near waterbodies and consist of open and semi-open areas with intermediate vegetation heights (SRD 2003). Vegetation cover and proximity to water are important for predator avoidance. Summer habitat may be found at greater distances (*i.e.*, 1 km to 2 km) from waterbodies in areas that have sufficient moisture, such as wet meadows, pastures, and drainage and irrigation ditches (Kendell 2002c). Northern leopard frogs avoid areas with little or no vegetation cover (*e.g.*, heavily grazed fields, mowed lawns or recently hayed or harvested fields) or areas with tall, dense marsh vegetation, grasses or extensive shrub or woody cover (Merrell 1977, SRD 2003). McAlpine and Dilworth (1989) noted that northern leopard frogs were found in vegetation with a mean height of 32 cm and a range of 9 cm to 85 cm. Dole (1965a) found that in fair weather in summer, northern leopard frogs were quite sedentary and spent the majority of the day sitting in "forms" created by clearing wet soil of dead vegetation. Northern leopard frogs will sit motionless waiting for prey items to come within range.

1.3.5 Area Requirements

Adult northern leopard frogs reportedly establish home ranges in the summer, with home ranges varying in size depending on habitat availability (Dole 1965b). Dole (1965b) studied northern leopard frog home ranges in two study areas in Michigan. In one study area, suitable habitat was more widespread and less confined and ponds were less readily available than in the second study area (Dole 1965b). Home ranges were larger in the first study area and increased in size with increasing frog body size, ranging from 283 m^2 for subadults, 362 m^2 for adult males, and 503 m^2 for adult females. In the second study area, home ranges varied from 68 m^2 for subadults, 78 m^2 for adult males, and 113 m^2 for adult females. Once adult frogs establish home ranges they reportedly return to the same region each summer (Dole 1965b). As described by Dole (1965b), home range size may be affected by several factors, including the degree of crowding, the amount of cover, the abundance of food, or the availability of moisture.

In the Cypress Hills, a YOY frog was found to travel a distance of 8 km from its natal pond over the course of two active seasons (Seburn *et al.* 1997). In general, YOY frogs in the Cypress Hills traveled up to 2.1 km downstream, and 1.0 km upstream (Seburn *et al.* 1997). Frogs dispersed over lesser distances (0.4 km) overland (Seburn *et al.* 1997). The majority of young frogs dispersed to ponds within 1 km within three weeks of metamorphosis (Seburn *et al.* 1997). The authors found that streams provide important dispersal corridors for northern leopard frogs in prairie environments. Seburn *et al.* (1997) suggest that aquatic routes may provide the required environmental cues and moisture to increase the number of days in which frogs can disperse. Ambient temperature and rainfall or humidity are other factors affecting dispersal of northern leopard frogs (Seburn *et al.* 1997). In Michigan, Dole (1971) reported that YOY frogs dispersed up to 800 m per night. He also recorded overland travel of up to 5 km by two adults. Based on known dispersal distances of 1.6 km to 2.0 km over the course of a single active season, northern leopard frogs in Alberta may occupy up to 12.6 km^2 at any given specific site, if suitable habitat is present (SRD 2003).

1.4 Northern Leopard Frog Response to Livestock Grazing

Intensive livestock grazing in riparian areas is most likely to have a detrimental impact on northern leopard frogs and their foraging, breeding and overwintering habitat (Wershler 1991, SRD 2003). Heavy grazing and trampling in riparian areas can reduce the quality of summer habitats by damaging foraging and shelter sites due to reduced vegetation cover and diversity, and creation of drier soil conditions (SRD 2003). Concentration of cattle around wetlands or along rivers not only reduces non-emergent bank vegetation, but can result in trampling or grazing of emergent vegetation and aquatic plants. This can alter and damage protective, reproductive and larval microhabitats and increase risk of predation on adult frogs and tadpoles (Jansen and Healey 2003). Trampling along the shore and in the water can also increase water turbidity and erosion, with possible negative consequences for tadpole and egg development (SRD 2003). Movements along the shoreline can additionally result in egg masses being dislodged or trampled causing direct mortality or harming egg development (SRD 2003). Didiuk (1999) reported that significant amphibian larvae mortality resulted from heavy livestock trampling of shoreline vegetation, high turbidity of water and poor water quality from cattle faeces in wetlands in the SNWA. Other potential impacts include deposition of livestock faeces and urine in overwintering ponds that can cause increased nutrient levels and an increased likelihood of anoxic conditions leading to winterkill (SRD 2003).

Elsewhere, Schmutzer *et al.* (2008) reported that wetlands grazed by cattle in Tennessee had lower amphibian diversity, tadpole abundance and water quality compared to wetlands not grazed by cattle. In contrast, Bull and Hayes (2000) found no significant difference in the number of egg masses or recently metamorphosized Columbia spotted frogs (*Rana luteiventris*) between grazed and ungrazed ponds in Oregon.

Impacts of grazing on wetlands and their associated frog communities are largely dependent on the level of grazing intensity. A study in an Australian floodplain revealed that frog communities, species richness and individual species of frogs declined with increased grazing intensity, as did the wetland condition (*i.e.*, water quality and aquatic vegetation components) (Jansen and Healey 2003). Reduced stocking rates and rest periods from grazing were suggested to improve wetland condition and increase frog production.

As northern leopard frogs co-existed with bison throughout much of their historical range in North America, this species is adapted to periodic disturbance, and appears to require some degree of herbivory to maintain intermediate vegetation heights and to create openings in dense cover that can be used for basking or foraging (Kendell pers. comm.). Therefore, complete exclusion of livestock from riparian habitats may improve the quality of habitat for northern leopard frogs in the short-term, but may have negative long-term effects (Kendell pers. comm.). Exclusion of livestock from riparian areas over the long-term leads to tall and dense vegetation cover in these areas that may impede the movement or foraging ability of northern leopard frogs or reduce the availability of suitable basking sites (Kendell pers. comm.). This scenario has been observed at the Prince's Spring area of the Remount Community Pasture in Alberta, a productive northern leopard frog breeding and overwintering site that has been fenced out from grazing since the late 1980's (Hofman, pers. comm.). Although northern leopard frog populations are still healthy at this site, limited, late fall grazing is being considered to selectively open up and

reduce vegetation density along the springs (Hofman pers. comm.). Fall grazing is also being considered as a tool to reduce the encroachment of tall, dense patches of reed canary grass (*Phalaris arundinacea*) in the Creston Valley Wildlife Management Area in British Columbia (Waye and Cooper 2001, Adama pers. comm.). Reed canary grass, a non-native species, has overgrown and displaced native vegetation along certain watercourses used by northern leopard frogs in this area. Northern leopard frogs appear to be excluded from areas with dense reed canary grass cover (Waye and Cooper 2001). Carefully managed cattle grazing in this instance may improve summer habitats and seasonal movements of northern leopard frogs.

Controlling the timing and intensity of livestock use in critical northern leopard frog habitats is important for maintaining or enhancing the quality of these habitats (Kendell pers. comm.). Grazing impacts should be restricted or minimized during sensitive periods near critical habitats. Heavy use early in the season (*i.e.*, April to May) is likely to have the most severe detrimental effects in terms of causing erosion or damaging breeding habitats and spawning micro-sites (Kendell pers. comm.). Mid-summer grazing (*i.e.*, June to July) may be less harmful as northern leopard frogs are more dispersed at this time of year in non-breeding areas and YOY have not yet emerged (Hofman pers. comm., Kendell pers. comm.). Grazing near breeding ponds in August may have an impact on the suitability and security of these sites for YOY frogs which emerge in large numbers at this time and stay close to breeding ponds. In September or October, grazing effects near breeding ponds are less of a concern unless these areas are also used for overwintering (Kendell pers. comm.). Grazing use near overwintering waterbodies early in the fall is not desirable as frogs congregate at these sites at this time of year. Winter grazing of riparian areas (*i.e.*, during frozen ground conditions), where feasible, may have the least detrimental consequences to northern leopard frogs and could be used to maintain suitable vegetation structure (Kendell pers. comm.). However, winter grazing intensity should be carefully managed. Heavy grazing during the dormant season can progressively set back woody species that play an essential role in stabilizing stream banks (Fitch and Adams 1998).

1.5 Grazing Systems and Northern Leopard Frog Habitat Management

Table III-15 provides an overview of six grazing systems and their potential positive or negative effects on northern leopard frogs and their habitat. A grazing system is a tool used to control the spatial distribution, timing, intensity, and frequency of livestock grazing (Holechek *et al.* 2003). Applied research is needed to properly assess the effects of various grazing systems on northern leopard frogs in the Milk River and South Saskatchewan River watersheds.

Table III-15 Grazing Systems and Northern Leopard Frog Habitat Management

Grazing System	Discussion
Continuous (Season-Long) Grazing	
Advantages:	Continuous grazing can be compatible with maintaining northern leopard frog habitats if stocking rates are low and stock water is available away from critical northern leopard frog habitats to minimize use in these areas.

Disadvantages:	Continuous grazing can be particularly damaging to riparian areas as livestock will congregate and linger near water throughout the duration of the grazing season from spring to fall (Ohmart 1996, Fitch and Adams 1998, Holechek *et al.* 2003). For this reason, continuous grazing nearly always results in overuse of riparian areas (Ohmart 1996). Heavy, persistent grazing in riparian areas can result in trampling and removal of vegetative cover, increased erosion and degraded water quality. This can have a negative impact on critical northern leopard frog overwintering, breeding and foraging habitats and can result in diminished northern leopard frog reproductive success or survival (SRD 2003). The degree of damage to northern leopard frog habitats under continuous grazing depends on grazing intensity, the effectiveness of grazing distribution techniques and the availability of alternate stock water sources away from critical habitats. Continuous grazing effects are generally more severe in well- developed riparian areas in complex rangeland landscapes and in permanent versus ephemeral riparian areas (Fitch and Adams 1998).
Rotational Grazing - Deferred Rotation	
Advantages:	Deferred spring grazing allows for deferral of grazing or trampling disturbance near riparian areas during periods when streambanks are saturated with moisture and are therefore most susceptible to erosion. Reduced erosion will help to improve water quality for northern leopard frog egg and tadpole development. Deferred spring grazing also minimizes impacts to aquatic or emergent vegetative cover during the critical northern leopard frog breeding period. Aquatic and emergent vegetative cover is important spawning habitat as it provides a surface for egg masses to attach to and offers shelter from predators. Deferred use during the spring has the added benefit of reducing risks of direct mortality to adults or egg masses from trampling.
Complementary Grazing	
Advantages:	Complementary grazing, a form of rotational grazing, allows for deferred grazing of native prairie, with tame pasture grazed earlier in the season when introduced forages are most nutritious and native grass are most vulnerable. As discussed above, deferral restricts cattle use of critical breeding habitats during the most sensitive spring period.
Disadvantages:	Complementary grazing requires the use of tame pasture which may not be available in all grazing systems. Tame pasture is often not suitable in many places in the mixedgrass or dry mixedgrass prairie due to the fragility of the soils and dry conditions. Tall and dense cover of introduced forages may impede northern leopard frog dispersal or diminish the quality of foraging or dispersal habitats (SRD 2003). Pasture rejuvenations (*i.e.*, re-seeding) can be expensive and may result in the cultivation of northern leopard frog breeding habitat, including temporary/seasonal wetlands as well as the

	margins of semi-permanent/permanent wetlands. If implementation of a complementary grazing system requires the conversion of native prairie to tame pasture it is not likely to provide added benefit to Alberta herpetiles.
Rotational Grazing	
Advantages:	Rotational grazing systems including rest- or deferred-rotation grazing are considered beneficial to restoring or maintaining the health of riparian areas (Fitch and Adams 1998). Rest-rotation grazing systems incorporate one or more years of rest into the grazing rotation. This allows for non-use during the growing season, permitting the recovery of degraded riparian areas by facilitating the growth and regeneration of herbaceous and woody vegetation. Deferred-rotation grazing allows the riparian area to be periodically rested and grazed later in the season. Introducing rest periods and deferring use during the critical spring period allows for improved vegetation cover and bank stability, reducing sediment loads entering the water. This subsequently reduces trampling risks to egg masses during the spring, offers improved foraging and shelter micro-sites for northern leopard frogs during the summer and may improve water quality. An overall improvement in water quality from lesser amounts of faecal contamination reduces the risk of anoxic conditions over the winter in northern leopard frog overwintering habitats.
Disadvantages:	The benefits of rotational grazing systems not only depend on timing grazing use of critical habitats to coincide with less sensitive periods, but are also dependent on stocking rates. At heavy stocking rates, impacts to critical habitats will increase and longer periods of recovery may be required. Reduced cover in upland areas due to heavy stocking rates also diminishes predator escape cover for dispersing northern leopard frogs and leaves them more prone to desiccation (Kendell pers. comm.). Rotational grazing that results in higher livestock densities in a given field could lead to greater trampling danger to frogs if grazing occurs during the breeding season or during dispersal to overwintering ponds.
Rotational Grazing – High Intensity Grazing	
Advantages:	Intensive, short periods of grazing in the fall may be beneficial in terms of reducing encroachment of tall exotic grasses such as reed canary grass in northern leopard frog foraging areas or dispersal corridors (Waye and Cooper 2001).
Disadvantages:	Most intensive grazing systems involve high stocking rates and high rates of forage utilization followed by periods of rest. The impact of this type of grazing system on northern leopard frogs can be severe if use is concentrated near breeding ponds, foraging habitat or waterbodies used for overwintering. Under high stocking rates, risks of dislodging or trampling egg masses are increased and there is a greater potential to reduce water

	quality due to erosion, intense removal of vegetation cover and high amounts of faecal contamination. Northern leopard frogs typically avoid heavily grazed areas when foraging and require an abundance of aquatic plants, emergent vegetation and moisture-tolerant grasses to provide optimal spawning habitat (Kendell 2002b).
Riparian Area Grazing	
Advantages:	As the goal of riparian area grazing systems is to improve the overall health of riparian areas, these systems have obvious benefits to northern leopard frogs. Improved vegetation cover, water quality and sediment trapping ability are some of the positive outcomes of a well-managed riparian area (Fitch and Adams 1998). These factors will improve the quality of northern leopard frog breeding, foraging and overwintering habitat.
Disadvantages:	Regeneration of woody vegetation and recovery of heavily degraded riparian areas may require lengthy periods of rest. For example, Homyack and Giuliano (2002) found that reptiles and amphibians in Pennsylvania may require more than four years to respond to streambank fencing. In prairie ecosystems, herbaceous plant communities that are dominated by exotic invasive graminoids will thrive with exclusion from grazing and are unlikely to revert back to a native plant community. Common invasive graminoids along riparian channels in southern Alberta include smooth brome (*Bromus inermis* ssp. *inermis*) and timothy grass (*Phleum pratense*). The impact of these exotic species on northern leopard frog foraging or dispersal behaviour requires further study.

1.6 Beneficial Management Practice Recommendations

In order to meet the annual requirements for all life history stages, the northern leopard frog requires a mosaic of critical habitat types for overwintering, breeding, and foraging (SRD 2003). These critical habitat types should be monitored, protected and managed properly. Landscape connectivity should also be maintained to ensure that northern leopard frogs can move between habitats (Pope *et al.* 2000). Facilitating dispersal is important to stabilize populations at a larger metapopulation scale, recolonize formerly extirpated sites and to maintain genetic connections between extant populations (Seburn *et al.* 1997, Pope *et al.* 2000). It is therefore important not to focus on simply protecting key habitats as isolated units, but rather to promote northern leopard frog recovery efforts by considering the constraints of the entire landscape (Pope *et al.* 2000).

The following general land-use and grazing recommendations provide a variety of means by which to protect or enhance northern leopard frog habitats. These recommendations apply to northern leopard frog populations within the Milk River and South Saskatchewan River watersheds and throughout the Grassland Natural Region of Alberta. Further research is required to improve our understanding of northern leopard frog ecology and their response to land-use

activities, such as livestock grazing, in the Milk River and South Saskatchewan River watersheds (Section 1.7).

1.6.1 General Recommendations

Native Prairie Conservation

- Conserve remaining native grassland habitat in the Milk River and South Saskatchewan River watersheds. Northern leopard frogs often disperse away from breeding wetlands during the summer foraging season (SRD 2003). The majority of northern leopard frogs observed during the 2005 provincial inventory were located in native prairie and "active pasture" as opposed to shrubland, forests, cropland or urban areas (Kendell *et al.* 2007). Avoid cultivation of native prairie habitats that connect or contain suitable northern leopard frog breeding, foraging or overwintering habitats. Cultivation not only results in loss of suitable habitat by draining wetlands and disturbing upland foraging sites, but also fragments and isolates remaining habitats, possibly impeding northern leopard frog dispersal and gene flow. Conserving native grassland will also benefit many other species at risk in southern Alberta.
- Support and fund not-for-profit groups such as ACA and the Nature Conservancy of Canada who undertake conservation initiatives on private land in southern Alberta (and elsewhere).
- Reclaim or restore previously cultivated or disturbed areas to native grassland, or less preferably, tame pasture (perennial cover). Balas *et al.* (2012) found that northern leopard frog occupancy probability was higher in seasonal wetlands surrounded by native prairie compared to seasonal wetlands surrounded by either perennial cover or cultivation, and that occupancy probability was higher in seasonal wetlands surrounded by perennial cover compared to cultivated seasonal wetlands.

Wetland Conservation and Restoration

- Reclaim drained wetlands that provided former northern leopard frog habitat and protect existing wetlands from being drained, plowed or structurally disturbed. Up to 70% of wetlands in Alberta's prairie pothole region have been converted to agricultural or other human-modified landscapes since European settlement (Locky 2011; GoA 2009; National Wetlands Working Group 1988; Schick 1972). Protecting buffer zones around wetlands and maintaining connectivity between different types of wetlands that form part of the habitat mosaic used by northern leopard frogs over the year is important.
- Protect key northern leopard frog breeding ponds, foraging habitats and overwintering sites from disturbance and monitor their condition (Seburn *et al.* 1997, SRD 2003). During the 2005 provincial northern leopard frog inventory, 63% (49 of 76) of active sites were described as threatened and 12% (6 of 49) had multiple threats (Kendell *et al.* 2007). The most common threat identified was cattle use (34%, or 26 of 76 sites), followed by human (16%, or 12 of 76 sites) and natural (14%, or 11 of 76 sites) threats.

- Ensure adherence to the Alberta Wetland Policy (GoA 2013b). Wetlands in Alberta are protected from disturbance under the Policy, which stresses a hierarchical approach to wetland disturbance, with avoidance as the preferred approach followed by minimization and then replacement. The Policy applies to all activities that have the potential to impact wetlands, whether carried out by a landowner or an energy company. The Policy applies to all natural wetlands in the province, including marshes and shallow open water wetlands, as well as all restored wetlands and wetlands constructed as part of wetland replacement initiatives.
- Avoid implementing water stabilization projects in northern leopard frog breeding waterbodies (ESRD 2012). Such projects could cause dislodging of northern leopard frog egg masses or reduced vegetation cover along the margins of wetlands.

Pest Control

- Avoid the use of pesticides or herbicides or chemical nitrogen-based fertilizers in the vicinity of breeding ponds or other key northern leopard frog habitats (Quellet *et al.* 1997, Christin *et al.* 2003). Atrazine, the most commonly used herbicide in the United States and possibly the world, is a known endocrine disruptor in northern leopard frogs, causing gonadal abnormalities and hermaphroditism in males, thereby potentially interfering with their ability to reproduce (Hayes *et al.* 2003). Atrazine is also an immune disruptor in northern leopard frogs, causing suppression of the immune system and making individuals more susceptible to pathogens (Brodkin *et al.* 2007). Paetow *et al*. (2012) found that while glyphosate and atrazine did not increase the susceptibility of northern leopard frogs to chytrid fungus infection, the herbicides did cause reduced growth, which could ultimately affect northern leopard frog survival. Rohr *et al*. (2008) found evidence that both atrazine and phosphate fertilizer increase exposure and susceptibility of northern leopard frogs to parasitic flatworms (larval trematodes) by suppressing frog immunity.
- Use biological control agents as opposed to insecticides to control mosquitoes in urban wetlands if northern leopard frogs and other amphibian species of concern are known to be present.
- Prevent the introduction of native and non-native fish species (*e.g.*, common carp) into northern leopard frog breeding and overwintering ponds, lakes and wetlands (SRD 2003). Fish stocking is considered a threat to northern leopard frog survival in Alberta (ANLFRT 2005). Northern leopard frogs normally breed in fishless ponds and likely have no natural defenses against predation by introduced fishes (Merrell 1977, Smith and Keinath 2007).
- Prevent the introduction of non-native plant species into northern leopard frog breeding and overwintering ponds. Introduced, invasive plant species such as purple loosestrife (*Lythrum salicaria*), reed canary grass and Eurasian milfoil (*Myriophyllum spicatum*) may alter the structure of wetland environments, negatively impacting northern leopard frogs (McAllister *et al.* 1999).

Habitat Protection and Industrial Development Mitigation

- Abide by AEP set-back guidelines and timing restrictions for northern leopard frog breeding wetlands in the Grassland and Parkland Natural Regions (GoA 2011). AEP recommends a year-round set-back of 100 m from northern leopard frog ponds for all levels of industrial developments (*i.e.*, low, medium and high), including land surveying, seismic drilling, and road, pipeline and facility construction. Where possible, to minimize potential impacts to northern leopard frogs, pipelines near northern leopard frog ponds should be constructed during frozen ground conditions in the winter. If construction occurs during the summer, trenches should be regularly checked for northern leopard frogs.
- Conduct pre-development wildlife surveys to locate northern leopard frog ponds in areas with suitable habitat. Ensure wildlife survey data is entered into the Fish and Wildlife Management Information System (FWMIS) database maintained by AEP.
- Avoid the creation of new dams or reservoirs along rivers and creeks known to support northern leopard frog populations. Dams and reservoirs often impede natural seasonal inundation of floodplain wetlands and can therefore negatively affect northern leopard frog breeding habitats and diminish reproductive success. Dams also result in direct loss of habitat for northern leopard frogs through reservoir flooding. New dams could also lead to loss of native prairie habitat due to the incentive to create additional irrigated cropland or hay fields.
- Where feasible, retain natural wetlands within new urban and residential developments.
- Use Protective Notations under the *Public Lands Act* to protect key northern leopard frog habitat.

Road Mortality

- Expand the provincial road mortality monitoring program to include amphibians to better determine mortality rates for northern leopard frogs in Alberta. Although road mortality does not appear to be a limiting factor in Alberta (ANLFRT 2005), traffic-induced mortality is a concern in other jurisdictions, including British Columbia and Ontario (Adama and Beaucher 2006, Eigenbrod *et al.* 2009). Although northern leopard frogs slow their movements around roads, they do not strongly avoid roads or traffic and can suffer high mortality as a result (Bouchard *et al.* 2009). Moreover, the amphibian strategy of remaining immobile while vehicles approach makes them especially vulnerable to traffic-induced mortality (Mazerolle *et al.* 2005).

Public Awareness and Education

- Establish and maintain good relationships with landowners in southern Alberta whose land contains species at risk. Work with and support groups such as MULTISAR, Cows and Fish and others to increase awareness of the importance of permanent waterbodies to northern leopard frogs among landowners in the province. Permanent and semi-permanent waterbodies are sometimes drained to provide additional cropland in agricultural areas. Permanent and semi-permanent waterbodies provide important breeding and/or overwintering habitat for northern leopard frogs (Cottonwood Consultants 1986, Kendell 2002c, SRD 2003). Conservation of permanent wetlands will

also result in other ecosystem benefits such as flood attenuation and control (Adamus and Stockwell 1983). Landowners should also be informed of their obligations under the Alberta Wetland Policy regarding wetland disturbance.

- Continue public information and education campaigns to promote awareness about the northern leopard frog and to encourage support in maintaining and monitoring northern leopard frog populations and their habitats. Extension efforts completed to date, including the 'Wanted: Northern Leopard Frog' poster (2006, revised in 2011) and the Amphibians on My Land (2011) stewardship brochure for the ranching and farming communities (Kendall, pers. comm.), are great efforts toward increasing awareness of species at risk in Alberta. Other extension activities completed to date include the publication of several magazine and newspaper articles on northern leopard frogs, interpretive sign installation and presentations to naturalist groups and members of the public (ESRD 2012). Additional extension work in the agricultural community could include skills development for wetland identification and delineation. Ephemeral, temporary and seasonal wetlands, as well as the margins of semi-permanent and permanent wetlands, are often cultivated when they are situated in crop and hay fields. If landowners are able to better identify wetlands (including the drier margins) they will be better able to avoid them during agricultural activities.
- Provide compensation to landowners who undertake habitat enhancement measures for wildlife species of concern on their land. Provide public acknowledgement of landowners who work cooperatively with wildlife managers to conserve species at risk. Be willing to listen to the concerns of landowners.
- Continue to support and fund programs aimed at preventing the introduction of aquatic and terrestrial invasive species into the province. The Alberta Invasive Plant Council produces a monthly e-newsletter (The Invader), writes fact sheets about invasive species and organizes an annual conference dedicated to non-native species issues in Alberta. Government programs include training for Conservation Officers and Parks Services Rangers in aquatic species detection and paid full-time Aquatic Invasive Species detectors, among others.
- Continue education efforts to inform the public about the harms of collecting northern leopard frogs for personal or commercial use.
- Continue efforts to promote education and understanding in the agricultural community of manure management, runoff and potential effects of livestock faeces on northern leopard frogs. In areas where residential development (recreational or otherwise) may come into conflict with northern leopard frog habitat management, work to increase awareness among owners of the negative effects of fertilizers and pesticides on northern leopard frogs.

Population Monitoring and Management

- Continue to refine northern leopard frog population size and trend estimates in southern Alberta. Three types of northern leopard frog surveys are ongoing in Alberta, including annual spring surveys to identify breeding site, late summer surveys of known breeding sites and targeted surveys for suspected new northern leopard frog sites or at historical sites (ESRD 2012). Northern leopard frog surveys should follow the ESRD Sensitive Species Inventory guidelines (GoA 2013a). Late summer surveys are preferable to spring

surveys (ESRD 2012). However, because of asynchronous breeding and variable development rates, multiple surveys over the course of the breeding season are necessary to accurately assess breeding habitat (Randall *et al.* 2014). Consider reporting 'zero data' when areas were searched for northern leopard frogs but no animals were found. Also, consider including photographic records with data submissions. Submit all survey results for inclusion in the FWMIS database.

- Continue with translocation efforts to reintroduce northern leopard frog populations in suitable habitat within its former range. Follow the guidelines for reintroduction outlined in the Alberta Northern Leopard Frog Reintroduction Strategy (Kendell and Prescott 2007). As previously discussed, there have been a number of successful northern leopard frog reintroductions in different watersheds in central and southern Alberta. However, translocation efforts have not included the Milk River or South Saskatchewan River watersheds to date as these were deemed to be lower priority watersheds for reintroductions (Kendell and Prescott 2007, ESRD 2012, ACA 2013). A study of the genetic diversity of northern leopard frogs in Alberta was completed in 2009 to identify potential source populations for reintroduction (Wilson *et al.* 2009).Wilson *et al.* (2009) make the following recommendations from a genetic perspective for northern leopard frog reintroductions in Alberta:

    - Where feasible, choose source populations that are geographically close to the reintroductions sites. Source populations from the same general area as the reintroduction site are more likely to be adapted to the local environmental conditions.
    - Source populations do not need to be from the same watershed as the reintroduction site. Straight line distances are equivalent to watershed-based distances.
    - The choice of source populations within the province may depend on availability and environment. If nearby source populations are unavailable, there are no obvious areas in the province that would serve as ideal source populations from a genetic standpoint.
    - Choose eggs from multiple masses in order to maximize genetic diversity at each reintroduction site.
    - Manitoba and Ontario may offer good source populations for northern leopard frogs due their high genetic diversity.
    - Monitor genetic diversity among new founder populations to ensure diversity and fitness are maintained.
    - Continue with translocation efforts to reintroduce northern leopard frog populations in suitable.
- Continue efforts to develop an effective and captive-rearing program for northern leopard frog reintroduction in Alberta (ESRD 2012).

1.6.2 Grazing Recommendations

Grazing systems that result in an improvement in wetland condition or improved riparian health will likely benefit northern leopard frogs. Heavy livestock use near riparian areas can be particularly harmful in the spring when banks are saturated and are susceptible to erosion and

northern leopard frogs are concentrated at breeding sites (SRD 2003, Wyman *et al*. 2006). It is particularly important to monitor and manage grazing impacts near permanent waterbodies such as spring-fed wetlands and along oxbows and meander scar wetlands in river floodplains that may provide overwintering and breeding habitat. Minimizing trampling along shorelines, maintaining vegetation cover (including aquatic and emergent vegetation) and improving water quality will benefit northern leopard frogs (SRD 2003, ANLFRT 2005). Managed properly, grazing can be compatible with maintaining and possibly improving northern leopard frog habitat. Appropriate grazing systems for northern leopard frogs should be developed site-specifically for participating ranches in southern Alberta conservation programs.

The following recommendations consider grazing management practices that can be used to improve the quality of northern leopard frog habitats:

- Use low to moderate stocking rates in pastures that contain or connect critical northern leopard frog breeding, foraging and overwintering habitats (Didiuk 1999). Adjust stocking rates during drought conditions.
- Observe utilization rates of 25% to 50% for native prairie-dominated fields to ensure adequate carry-over of residual vegetation and litter (Adams *et al.* 2005, 2013a, b). This will help to prevent uniform grazing effects and retain sufficient upland cover for northern leopard frogs dispersing overland. Sufficient cover offers shelter from predators and helps to prevent desiccation of frogs.
- Observe conservative utilization rates (25% to 50%) in riparian habitats where northern leopard frogs occur to ensure that vegetative cover remains intact (Fitch and Adams 1998).
- Place salt or mineral sources at least 1 km from natural water bodies, where possible (Adams *et al.* 1986). Placing salt away from water forces cattle to make better use of the range and reduces the amount of time cattle spend at water sources.
- Provide alternate watering sites (*e.g.*, off-site watering systems) to reduce impact to sensitive northern leopard frog ponds, streams or other natural riparian habitats. Cattle have been shown to prefer drinking from a water trough and will travel further to a trough rather than drink from a stream or wetland/pond when given free access to both (Surber *et al.* 1996, Veira and Liggins 2002, Veira 2003). Off-site watering can provide a more reliable and cleaner water source for cattle. Improved water quality can improve livestock health and weight gain (Veira 2003). Install wildlife escape structures (*e.g.*, ladders, ramps) to ensure wildlife do not get trapped in water troughs (Kendall pers. comm.). Several off-stream watering projects, including solar-powered and gravity-fed systems, have already been implemented by ACA with different landowners in the province (Kendall pers. comm.).
- Off-site watering systems and salt placement are particularly effective tools in relatively flat prairie; however, in variable terrain, additional methods (as described below) may be required to improve cattle distribution and minimize impact to critical northern leopard frog habitats.
- Provide graveled or hardened access points for livestock at select points along a stream or river away from productive northern leopard frog breeding or overwintering habitats.

This focuses cattle use at a few predetermined locations and reduces sedimentation of watercourses.

- Use rest- or deferred-rotation grazing systems to ensure sufficient periods of rest and recovery in riparian areas.
- Defer livestock use near riparian areas during vulnerable periods in the spring when stream banks are soft and are susceptible to slumping and erosion. Heavy trampling in the spring poses a particular threat to spawning adult northern leopard frogs, egg masses, and tadpoles in breeding habitats.
- Avoid heavy impacts to breeding ponds in August to prevent trampling and conserve vegetation cover for emerging YOY frogs.
- In areas with distinct upland and lowland areas, explore the use of riparian pastures to defer use in riparian areas during the spring. Riparian pastures are created by fencing upland terrain and riparian landscape units separately (Fitch and Adams 1998).
- Fence out select riparian areas (such as spring-fed overwintering streams or breeding ponds) to restore water quality or vegetation cover or to prevent damage at key times of the year (*i.e.*, during breeding). Monitor fenced out sites to ensure that vegetation growth does not exceed the threshold required for northern leopard frog foraging or movement. Undertake vegetation management actions (*e.g.*, targeted, periodic grazing) to maintain optimal vegetation cover for northern leopard frogs.

1.7 Research Recommendations

Extensive research has been conducted on northern leopard frogs in Alberta by organizations such as ACA. Ongoing monitoring is needed to assess the status of northern leopard frogs in Alberta and to evaluate the long-term success of northern leopard frog reintroduction efforts. ACA is currently monitoring the success of several reintroduction sites in the Milk River and South Saskatchewan River watersheds as well as elsewhere in the province (ACA 2010, 2013). Monitoring of the success of the reintroduction efforts will allow for refinement of the existing protocols, which could lead to further successes (ESRD 2012).

Key research needs for the northern leopard frog include:

- A better understanding of northern leopard frog overwintering habitat site selection (ESRD 2012). A better understanding is needed of what constitutes ideal winter habitat. This knowledge will aid reintroduction and wetland conservation efforts.
- Conduct field validation of the HSI and RSF models developed for northern leopard frog habitat use in southern Alberta (Stevens *et al.* 2010, Gan *et al*. 2014). Preliminary field validation of the HSI model led to the discovery of 11 new northern leopard frog sites out of 23 sites searched (ESRD 2012). Refine the HSI and RSF models as new data becomes available.
- Continue monitoring and research into the prevalence and spread of Chytrid fungus (*Batracochytrium dendrobatidis*) in Alberta northern leopard frog populations and its potential role in population declines (ESRD 2012).

- Develop population viability models to assess the long-term viability of northern leopard frog populations in southern Alberta under existing climatic conditions and under varying management and land-use scenarios (ESRD 2012).
- Determine the effectiveness of various riparian area and other grazing systems and strategies in terms of their benefits to northern leopard frogs. In particular, the effects of winter grazing on northern leopard frog habitat should be studied. The role of grazing in maintaining summering or dispersal habitats for northern leopard frogs also requires study. To assess the impact of grazing on summering habitats, research is first needed to further characterize northern leopard frog summering habitats and dispersal patterns in the Milk River and South Saskatchewan River watersheds and to assess the impact of herbaceous exotic species invasion. Northern leopard frog dispersal studies may provide insight into mitigation measures that are needed to reduce landscape barriers from human activities and restore habitat connectivity. Collectively, these types of studies will help to refine land-use and grazing management recommendations for enhancing or protecting northern leopard frog habitats. The MULTISAR program may be a good way to research the effectiveness of different grazing systems/strategies in benefiting northern leopard frog habitat.
- Investigate the long-term implications of drought and climate change on northern leopard frog survival (Environment Canada 2013).

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IV: OTHER

A. WEIDEMEYER'S ADMIRAL

1 INTRODUCTION

The purpose of this report is to summarize the ecology and habitat requirements of the Weidemeyer's admiral (*Limenitis weidemeyerii*) in southern Alberta. Based on this information and supporting scientific studies, the potential effects of livestock grazing on this species and its habitats is assessed. This discussion is followed by a summary of recommended beneficial management practices to enhance habitat for Weidemeyer's admirals in the Milk River Watershed in Alberta. Lastly, a brief review of key research recommendations is provided.

1.1 Background

The Weidemeyer's admiral is a distinctive, large black butterfly with bold white bands on both wings and greyish white markings on the underside of the hind wings (Kondla 2000). This species is widely distributed in the western interior of the United States, but is limited in its distribution in Canada. The only known resident population in Canada occurs along an estimated 80 km stretch of the Milk River and its tributaries in southern Alberta (Kondla 2000). Riparian deciduous forests, shrubby vegetation along river valleys and coulees, and spring and seepage sites provides key habitat for the species.

The Weidemeyer's admiral is listed as 'May Be At Risk' in Alberta under the General Status listing (Government of Alberta [GoA] 2016). This species is listed as a species of 'Special Concern' under the provincial *Wildlife Act*. At the national level, this species was designated as a species of 'Special Concern' in 2012 by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC), and its designation remains unchanged to this date (COSEWIC 2016). The Weidemeyer's admiral has not been assessed internationally by the International Union for Conservation of Nature and Natural Resources (IUCN 2016).

The population of Weidemeyer's admirals in Alberta is estimated to be between 1,800 and 3,200 individuals, with only seven to 11 known records (populations) (Kondla 2000; Alberta Sustainable Resource Development (SRD) and Alberta Conservation Association (ACA) 2005, GoA 2016). This population estimate may be generous based on the limited habitat available to the species. Weidemeyer's admiral populations are patchy and discontinuous, reflecting the nature of suitable habitat patches (Kondla 2000).

Admiral butterflies are naturally limited by factors such as predation, weather events and the availability of suitable habitat. Human-induced limiting factors include recreational or scientific collecting and habitat destruction due to agriculture or industrial development (Kondla 2000). Overall, these anthropogenic factors are considered to have a minor influence on admiral populations (Kondla 2000). Most admiral habitat patches are not suitable for mechanized agricultural tillage, while developments such as pipeline construction usually only affect small habitat patches (Kondla 2000). Intensive livestock grazing in admiral habitats is considered among the more significant anthropogenic threats due to the potential for more widespread impacts (Kondla 2000). Intense livestock grazing can degrade the quality of admiral butterfly

habitat in riparian forests and coulee bottoms and may result in incidental mortalities through foraging and trampling (SRD and ACA 2005, GoA 2016).

1.2 Ecology

Little published information is available regarding the biology of the Weidemeyer's admiral (Kondla 2000). In Alberta, admirals have a one-year generation time, with the wintering stage being third instar larvae in a hibernaculum (Kondla 2000). Adults have been observed in flight from mid-June to mid-July (Kondla 2000). Males are thought to be territorial as they participate in perching and patrolling types of mate-locating behavior as well as contest fighting (Rosenberg and Enquist 1991, Kondla 2000). Males are usually more visible than females, which spend most of their time in shrubs (Kondla 2000). Females lay grayish-green eggs singly on the upper side of leaf tips of host plants (Kondla 2000). Common host plants include saskatoon (*Amelanchier alnifolia*), aspen (*Populus tremuloides*), willow (*Salix* spp.) and wild cherries (*Prunus* spp.) (Kondla 2000). In Alberta, only saskatoon has been confirmed as a substrate for egg-laying (SRD and ACA 2005). Admiral caterpillars feed on the leaves of these plants prior to hibernating through the winter (Kondla 2000). Adults emerge the next summer following pupation, mainly in mid- to late June, with some surviving until August (Acorn 1993).

Weidemeyer's admiral hybrids can occur with other admiral species or sub-species where their ranges meet (Kondla 2000). It is believed there is extensive genetic exchange between populations in Canada and the United States (Kondla 2000).

1.2.1 Diet

Admiral caterpillars feed on the leaves of deciduous tree and shrub host plants as described previously. Adult butterflies feed mainly on tree sap, carrion and flower nectar from plant species such as western clematis (*Clematis ligusticifolia*) (Kondla 2000). Moisture and mineral uptake is obtained from mud (Kondla 2000, SRD and ACA 2005).

1.2.2 Predators

Butterflies fall prey to numerous predators, including birds, other insects and spiders (Acorn 1993, COSEWIC 2012).

1.3 Habitat Requirements

1.3.1 General

Weidemeyer's admirals are associated with shrub complexes and woody riparian vegetation along the valleys of the Milk River and its tributaries, including spring and seepage sites and coulees (Kondla 2000). Deciduous trees and shrubby areas provide the necessary habitat components required by this species. Required habitat elements include: larval host plants, shelter from strong winds, flowering plants for nectar, damp soil or mud for mineral uptake and elevated perches for mate-location (Kondla 2000, SRD and ACA 2005). Within the Milk River Watershed, adult Weidemeyer's admirals have been found in association with cottonwoods (*Populus deltoides* and hybrids), saskatoon, western clematis and thorny buffaloberry (*Shepherdia argentea*) (Pike 1987). It has been concluded that there are three primary habitat

types for Weidemeyer's admirals in Alberta: riparian forest and shrub communities in the valley bottoms of the Milk River; pockets of trees and shrubs along tributaries in coulees and ravines; and small patches of choke cherry (*Prunus virginiana*) and saskatoon in coulees and ravines (SRD and ACA 2005).

An HSI model was developed for Weidemeyer's admiral habitat use in the Milk River Watershed as part of the Multiple Species at Risk (MULTISAR) conservation program (Taylor 2004). Because Weidemeyer's admirals are limited to valleys as well as to areas where shrubs are present, topography and shrub cover were selected as the two habitat variables in this HSI model. According to the model, the best potential habitat for these butterflies is in the valley shrub complexes along tributaries to the Milk River, where there have been known occurrences of this species. The abundance of potential habitat identified in the HSI model may be an overestimate because of the coarseness of scale of vegetation polygons and the inclusion of shrubs that are not larval host plants (SRD and ACA 2005, COSEWIC 2012).

1.3.2 Area Requirements

Weidemeyer's admiral has been found in very small habitat patches, but they use coulees and ravines as flight corridors and can access small patches throughout these systems. It is probable that they travel across open prairie, but this has not been confirmed. The average distance travelled by this species was 166 m in a flight season, but ranged from 0 m to 2850 m (SRD and ACA 2005). Greater study on the movement patterns and area requirements of Weidemeyer's admirals are needed.

1.4 Grazing and Weidemeyer's Admirals

Few studies have assessed the potential impacts of livestock grazing on the Weidemeyer's admiral. A brief overview of the potential implications of grazing on this butterfly species and its habitat will be provided here; however, due to the limited information available, a comparative table of various grazing systems will not be given.

Intensive livestock grazing in woody riparian sites along valleys can negatively impact suitable admiral butterfly habitat (Kondla 2000, SRD and ACA 2005). Intense trampling and browsing may degrade admiral habitat by reducing the quantity and quality of larval host plants or nectar sources (Kondla 2000, SRD and ACA 2005). Heavy use may also have direct impacts on admirals, such as dislodging larvae from host plants and causing incidental mortality of immature stages due to browsing of host plants (Kondla 2000, 2005). Local loss of admiral habitat has also occurred because of stock watering structures (SRD and ACA 2005). Consequently, livestock use in valley and coulee habitats may need to be managed and monitored to maintain the quality of these sites for Weidemeyer's admirals.

1.5 Grazing Systems and Weidemeyer's Admiral Habitat Management

Various grazing management strategies have been suggested to maintain or improve the health of riparian areas (Fitch and Adams 1998). A key consideration is to protect riparian areas from grazing during vulnerable periods, such as in the spring when banks are saturated and easily damaged, or in the fall when woody species are most vulnerable to browsing (Fitch and Adams

1998). Other considerations are ensuring appropriate levels of vegetation utilization and using distribution tools such as off-stream watering and salt placement (Fitch and Adams 1998). Deferred or rest-rotation grazing systems are generally preferred over season-long or continuous grazing, particularly in complex landscapes with well-developed riparian areas (Fitch and Adams 1998). Under rest-rotation grazing, grazing is rotated between multiple pasture units with at least one pasture receiving a full year's rest in a grazing cycle (Fitch and Adams 1998). Allowing areas to be rested helps to maintain woody species and restore fragile stream banks. For example, a known admiral butterfly host plant, saskatoon, is considered a highly preferred browse species for livestock and wild ungulates and will decrease under persistent heavy utilization (Thompson and Hansen 2002). Providing periods of rest will help to maintain healthy saskatoon communities, as this species is known to repopulate sites where it was previously severely reduced (Thompson and Hansen 2002). Similarly, periodic rest is necessary to maintain healthy cottonwood stands (Fitch and Adams 1998). Heavy browsing and trampling by cattle is known to negatively affect cottonwood regeneration and density along the Milk River (Bradley and Smith 1986). Where necessary, fencing may be required to allow the recovery of riparian vegetation in frequently heavily used areas (Fitch and Adams 1998).

1.6 Beneficial Management Practice Recommendations

Due to the potential for livestock grazing to impact the quality of riparian habitats, it is important to consider management strategies that aim to balance livestock use with sustained riparian health. It is also important to consider strategies to minimize potential impacts due to other human activities, such as industrial development or water diversion projects, which can pose a risk to Weidemeyer's admirals or their habitats. Maintaining healthy riparian habitats will help to retain key habitats for Weidemeyer's admirals, including shrubby valley bottoms, coulees and ravines associated with watercourses and draws (Kondla 2000, SRD and ACA 2005).

The following general land-use and grazing recommendations provide a variety of means by which to protect or enhance Weidemeyer's admiral habitats. These recommendations apply to the range of this species within the Milk River Watershed and throughout the Grassland Natural Region of Alberta. Further research is required (Section 7) to improve our understanding of these species in order to evaluate or refine appropriate management recommendations.

1.6.1 General Recommendations

- Avoid water diversion projects or the creation of new dams or reservoirs along the Milk River and its tributaries. Flooding from the creation of large reservoirs can eliminate suitable habitat for Weidemeyer's admirals. As river valley habitats are critical to this species, flooding can have potentially severe consequences. Damming or water diversion can also negatively affect riparian vegetation communities, such as cottonwood forests, by impeding seedling establishment (Bradley and Smith 1986). Cottonwoods are reliant on periodic flooding during periods of seed dispersal to provide suitable conditions for seedling establishment (Bradley and Smith 1986). Riparian cottonwood forests and associated plant communities, as discussed, provide the required habitat components for Weidemeyer's admirals in the Milk River Watershed (Pike 1987).

- The Milk River Watershed HSI maps developed for the Weidemeyer's admiral (Taylor 2004) should be used to inform environmental planning of development activities to ensure key habitats are avoided wherever possible. The HSI maps should be revised as additional information or refined GIS data layers become available. Resource Selection Function models should be substituted for HSI models if possible due to the greater statistical rigour of the former.
- At present there are no timing or set-back guidelines for industrial development activities from known critical Weidemeyer's admiral habitats (GoA 2011). Developing such restrictions may be of benefit to the species.
- Maintain or restore the quality of riparian areas and springs that provide key habitat for Weidemeyer's admirals.
- Avoid the use of pesticides or herbicides near tributaries used by Weidemeyer's admirals. The use of these chemicals may have direct adverse effects on admirals (Forsyth 1993, 1996).
- Implement invasive species monitoring and management of Russian olive (*Elaeagnus angustifolium*) and tamarisk/salt cedar (*Tamarix chinensis, T. ramosissima*) in the Milk River area, where applicable. There have been occurrences of Russian olive along the Milk River but no reported occurrences of tamarisk/salt cedar to date. When these non-native, invasive species establish they negatively impact native cottonwood stands and therefore have the potential to impact Weidemeyer's admiral habitat. It is important to detect the infestations early and to respond rapidly to eliminate and control these species before they have the chance to establish and spread (COSEWIC 2012).

1.6.2 Grazing Recommendations

Healthy riparian areas with a diverse shrub and forb community also provide key habitat for Weidemeyer's admirals. The timing and intensity of grazing, the ability to control animal distribution and access to water and the frequency and duration of rest periods affect how severely cattle impact riparian habitats (Fitch and Adams 1998). Appropriate grazing systems should be developed site-specifically for participating ranches as part of the MULTISAR conservation program in southern Alberta.

The following recommendations outline grazing management practices that can be used to improve the quality of riparian habitats for Weidemeyer's admirals:

- Observe conservative stocking rates in fields that contain riparian areas to maintain vigorous herbaceous and woody vegetation (Fitch and Adams 1998).
- Place salt or mineral sources at least 1 km from natural waterbodies, where possible (Adams *et al.* 1986). Placing salt away from water forces cattle to make better use of the range and reduces the amount of time cattle spend at water sources.
- Provide alternate watering sites (*e.g.*, dugouts or troughs) to reduce impact to natural riparian habitats. Cattle have been shown to prefer drinking from a water trough and will travel further to a trough rather than drink from a stream when given free access to both (Veira 2003). Off-site watering can provide a more reliable and cleaner water source for cattle and creates additional water sources for bats in an arid landscape. Improved water quality can also improve livestock health and weight gain (Veira 2003).

- Off-site watering systems and salt placement are particularly effective tools for managing livestock distribution and minimize impact to riparian habitats in relatively flat prairie; however, in variable terrain additional methods (as described below) may be required.
- Provide graveled or hardened access points for livestock at select points along streams and rivers away from critical Weidemeyer's admiral habitat. Hardened access points focus cattle use at predetermined locations and reduce sedimentation of watercourses.
- Use rest or deferred rotation grazing systems to ensure sufficient periods of rest and recovery in riparian areas (Fitch and Adams 1998). Providing periods of rest allows for the recovery of woody species that may be important host plants for admiral butterflies. Periodic rest also reduces potential incidental mortality of butterfly larvae due to trampling or browsing. Several years of rest may be required where the goal is to regenerate new trees like cottonwoods (Fitch and Adams 1998).
- Defer use near riparian areas during vulnerable periods such as in the spring when stream banks are saturated and are susceptible to slumping and erosion. Also, avoid heavy use of riparian areas in the fall when woody vegetation is most vulnerable to browsing (Fitch and Adams 1998).
- In areas with distinct upland and lowland areas, explore the use of riparian pastures to defer use in riparian areas during the spring. Riparian pastures are created by fencing upland terrain and riparian landscape units separately (Fitch and Adams 1998).
- Where necessary, fence out select riparian areas to restore water quality or vegetation cover or to prevent damage at key times of the year (*i.e.*, during spring and fall).

1.7 Research Recommendations

There is limited information available about the population status or ecology of Weidemeyer's admirals in southern Alberta (Kondla 2000). Further research is also needed to better evaluate the impact of land-use activities on this species in the Milk River Watershed.

Several key research recommendations are given below:

- Determine specific host or nectar plant species associations. Investigate the possibility of conducting larvae stage surveys to identify host plant species in Alberta (SRD 2012).
- Investigate the possibility of conducting mark-release-recapture studies or transect counts to monitor Weidemeyer's admiral population abundance and trends in the Milk River Watershed (Kondla 2000, SRD 2012).
- Research Weidemeyer's admiral biology, habitat use and seasonal movements.
- Determine the forage value or susceptibility of host or nectar plants to livestock versus wild ungulate herbivory.
- Assess the effectiveness of suggested riparian area grazing strategies for maintaining or improving the quality of Weidemeyer's admiral habitats.
- Evaluate the most appropriate timing and frequency of livestock grazing to minimize potential for incidental mortality of butterfly larvae and to sustain healthy communities of host or nectar plants.
- Investigate ways to control/eliminate Russian olive and tamarisk/salt cedar to ensure Weidemeyer's admiral habitat is not impacted (SRD 2012).

- Implement education and communication programs to promote Weidemeyer's admiral habitat conservation, with the inclusion of landowners, government, industry and the public (SRD 2012).

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SYNTHESIS

1 BENEFICIAL MANAGEMENT PRACTICES SYNTHESIS

Part III of this report provides an overview of the ecological and habitat requirements for 33 select management species within the Milk River and South Saskatchewan River watersheds. Beneficial management practices (BMPs) are presented for individual species or species groups. This information is intended to assist with the design of appropriate stewardship activities under the Multiple Species at Risk (MULTISAR) Conservation Strategy for southern Alberta.

There are several key recommendations that are repeatedly stressed for the majority of species and species groups. Of prime importance is the need for the conservation and maintenance of large, contiguous blocks of native prairie and unique landscape features or habitat types such as sand hills, badlands, river valleys and riparian areas. Conservation of natural corridors linking native prairie, wetland or unique landscape features is also important to facilitate dispersal and connectivity between wildlife populations. Maintaining the integrity of native habitats requires minimizing the potential effects of development or high-impact human activities, and avoiding loss of habitat due to cultivation or encroachment of invasive species. Another important consideration is the maintenance and management of key natural disturbance processes such as grazing within native habitats.

1.1 Key Land-use Recommendations

Cultivation, industrial and commercial development, road construction and water diversion projects have the greatest potential to fragment or remove valued habitats or critical habitat components for select management species. Undertaking pre-development environmental planning, applying minimal disturbance construction techniques and observing timing and set-back restrictions near important wildlife habitats can minimize impacts from industrial development. Complete exclusion of certain critical wildlife areas from industrial disturbance is also necessary, especially considering the cumulative impact of development activities and associated infrastructure. The value of existing cropland areas within the Milk River and South Saskatchewan River watersheds to wildlife species can be improved by careful attention to appropriate farming practices, such as minimum or zero tillage, delayed summer haying (to avoid the nesting season) and the use of organic farming techniques. Organic farming practices have numerous benefits for wildlife and for maintaining healthy ecosystems. Organic farming promotes the use of natural biological control methods for pest management as well as the use of natural versus synthetic fertilizers. Crop rotations, crop residues, animal manures, legumes and green manures and mechanical cultivation are used in organic farming as alternative methods to maintain soil fertility and productivity. In this way, organic farming emphasizes management procedures that work with natural cycles and processes to conserve beneficial soil organisms and natural pest controls. Multiple crop systems and avoidance of pesticide use creates increased diversity in vegetation structure and food items available to wildlife and may help to improve water quality in local watersheds. The use of pesticides or herbicides, organo-chlorine compounds in particular, has the potential to diminish food supplies for wildlife and can cause direct toxicity or reduced survival or reproductive success. Toxins are often passed through the food chain to higher order predators through the processes of bioconcentration, bioaccumulation and biomagnification. Moreover, many insect or rodent species targeted by pesticide campaigns play a vital ecological role as important prey species. Rodent “pests” such as the Richardson’s

ground squirrel have ecological importance not only as a crucial prey source but also for providing habitat for numerous wildlife species through its burrowing activities. Promoting predators as natural control agents and appropriate grassland management can be used as alternative methods of controlling ground squirrels within tolerable thresholds.

1.2 Important Grazing Management Principles and Recommendations

1.2.1 Maintaining a Range of Natural Variation

Livestock grazing can provide an economical and sustainable use of the landscape while allowing for the conservation of natural landscapes and native prairie. Grazing by herds of large ungulates in combination with fire and drought cycles are natural processes that shaped the evolution of prairie ecosystems for thousands of years. Bison were a keystone species of the North American Great Plains. Livestock replaced bison as the dominant grazers in southern Alberta by the late 1800's. Grazing by large ungulates has the potential to modify the composition and structure of plant communities. This has an indirect or direct influence on various wildlife species through effects on food sources and/or habitat quality. If appropriately managed, grazing can be used to enhance, maintain or create habitat for multiple species. In this respect, it is important to remember that certain rare, unique, threatened or endangered species occur on rangelands because of, and not in spite of, appropriate historic stewardship practices such as livestock grazing.

Managing for a "range of variation" in the size, intensity and return interval of disturbance processes is an important concept of prairie ecosystem management (Bradley and Wallis 1996, Adams *et al.* 2004). Prairie fauna and flora are adapted to a variety of cover and disturbance thresholds. For example, although migratory herds of bison may have had significant local impacts, periods of high impact would likely have been offset by periods of rest when local forage or water sources became depleted. Moreover, although some areas may have been subject to frequent bison disturbance, other areas may not have been exposed to bison for several decades (Bradley and Wallis 1996). The impact of bison also fluctuated in response to climate cycles, fires and patterns of forage quantity or quality, creating a range of cover types across the landscape. Wildlife species are consequently adapted to different intensities of grazing. Species such as the mountain plover and Richardson's ground squirrel occur in more heavily grazed grasslands, while others, such as Baird's sparrows and Sprague's pipits, prefer more lightly grazed areas. Species such as the burrowing owl require short vegetation near nest sites, but require taller cover within 600 m of nests to maintain suitable conditions for a stable small mammal prey base. Meadow voles, a key prey item for burrowing owls and short-eared owls, typically occur in areas with taller grass cover. Current livestock grazing systems may achieve a range of variation in vegetation cover and create habitat for a diversity of wildlife, if they are tailored to promote an interspersion of heavy, moderate and lightly grazed patches. The use of multiple grazing systems over the landscape, as opposed to a single system, is also beneficial for promoting heterogeneous cover types at larger scales. Range management that over-intensifies or homogenizes grazing across the landscape will tend to reduce the range of natural variation (Bradley and Wallis 1996).

1.2.2 Adaptive Management

To maximize the benefits to wildlife and the ecology of an area, grazing management must be flexible and custom-tailored to meet the needs and character of each management unit. For example, several grazing systems can be applied on a single ranching operation to take advantage of topographic features, multiple types of native range or the availability of tame pasture that may exist in an area. Designing an appropriate grazing management system, therefore, is ultimately dependent on local conditions and economic considerations balanced with other resource management needs and objectives (Adams *et al.* 2004). There is no universal 'best' grazing system that can be applied in all cases. The success of a grazing system depends on appropriate stocking rates, animal distribution, proper use and monitoring. Grazing systems need to be considered as part of an adaptive approach to landscape management to react to changes in climate, management priorities and resource conditions. An adaptive management approach involves goal setting, implementation of a planned strategy, and continual monitoring and adjustment. Before any type of management system can be implemented, the objectives of the ranching operation and wildlife habitat requirements should first be defined. Secondly, knowledge of the range resource base should be obtained (*e.g.*, present range and riparian health and distribution of critical wildlife habitat features). Landowner support and commitment to the implementation of a habitat conservation strategy is the over-riding factor to its success.

1.2.3 Grazing Management Tools and Principles

Beneficial livestock grazing practices consider grazing capacities and stocking rates (or grazing intensity), timing (*i.e.*, onset and duration) and frequency of grazing, distribution practices, the application of appropriate grazing systems, as well as specialized grazing practices (Adams *et al.* 2004). Of these factors, grazing intensity can often be a decisive factor in terms of potential impacts to wildlife. Grazing intensity affects the amount of forage biomass removed by livestock and the amount of residual cover that remains. This has an effect on forage resources and on the availability of nesting, predator escape or thermal cover available to wildlife. Grazing capacities and stocking rates also affects the degree of impact livestock have on riparian habitats. As discussed below, riparian areas provide important habitat for a diversity of wildlife. Range and riparian health, wildlife management priorities, forage productivity and the contribution of wild native grazers need to be considered when establishing appropriate grazing capacities. Light to moderate grazing intensity is generally considered beneficial to most wildlife species as it allows for a gradation of use across fields, and promotes the formation of overgrazed and undergrazed patches. Patch grazing contributes to landscape heterogeneity. Grazing systems can be used to create planned heterogeneity on a larger scale by controlling grazing pressure and timing within multiple fields. Controlling livestock distribution through the use of herding, salt or upland water placement is another mechanism to create planned heterogeneous use across the landscape. The class and age of livestock are other considerations with respect to improving livestock distribution. Most distribution tools work best in conjunction with grazing systems. Without effective livestock distribution, grazing is usually restricted to preferred areas. This not only has implications to wildlife habitat but it also reduces the effective grazeable land area and consequently leads to reduced stocking rates. Concentration of grazing in preferred areas may also lead to other range management concerns such as weed species encroachment and soil erosion.

Livestock distribution, timing and frequency can be designed not only to promote heterogeneity, but can also be coordinated with the life histories of select management species to avoid critical habitats during sensitive periods. For example, deferring grazing during the spring near sharp-tailed grouse leks minimizes the risk of trampling nests and eggs and avoids the removal of critical nesting or brood-rearing vegetation cover. Spring deferral has the added advantage of avoiding grazing during vulnerable growth phases of native plants and allows for improved seed production, seedling establishment and restored plant vigour. Deferral is usually recommended until after May 15 through to June 15. Spring deferral requires the use of complementary grazing or deferred rotation grazing. Complementary grazing involves the use of tame pasture early in the season. As conservation of native prairie is pivotal, this strategy is only recommended where tame pasture already exists within the grazing operation. Under deferred rotation grazing, spring use is alternated between multiple pastures from year to year.

To minimize potential impacts to wildlife, it is important to carefully plan the distribution of high use points associated with water or salting facilities, fences, corrals or winter feeding stations in relation to critical habitat components. These facilities should be set back from critical features such as leks, hibernacula, traditional nesting sites and key winter ranges. Fences and watering and livestock handling facilities not only concentrate livestock use, they also provide perches for avian predators and potentially serve as sources for invasion of exotic species.

1.2.4 Riparian Area Management

Although factors such as vegetation type, soils, slopes, terrain, insects and weather influence livestock distribution over the landscape, water is ultimately the most influential factor, particularly in semi-arid environments. Unlike bison, cattle tend to linger for longer periods near water sources and tend not to travel as far from water. Consequently, cattle can have significant impacts on natural floodplain and riparian habitats. The impacts of cattle on ephemeral wetlands and riparian habitats can influence numerous wildlife species, including northern leopard frogs, plains spadefoot and Great Plains toads, sharp-tailed grouse, loggerhead shrikes, ferruginous hawks, golden eagles, Swainson’s hawks, Weidemeyer’s admirals, mule and white-tailed deer. The structural complexity and plant species diversity associated with riparian habitats provides thermal and escape cover, forage and nesting material for these species. Over-use of riparian areas due to heavy trampling and/or browsing can result in denudation of bank vegetation, increased erosion and siltation of waterbodies and negative effects on wildlife.

Several strategies have been proposed to manage livestock use of riparian areas, while maintaining their structure and function (*i.e.*, their health) (Fitch and Adams 1998). A key consideration is to protect riparian areas from grazing during vulnerable periods, such as in the spring when banks are saturated and easily damaged, or in the fall when woody species are most vulnerable to browsing. Other considerations include ensuring appropriate levels of vegetation utilization (between 25% and 50%) and using distribution tools such as off-stream watering and salt placement. Deferred or rest-rotation grazing systems are generally preferred over season-long or continuous grazing to provide periods of rest. Under rest-rotation grazing, grazing is rotated between multiple fields with at least one field receiving a full year of rest in a grazing cycle. Allowing areas to be rested helps to maintain woody species and restore fragile stream banks. Where necessary, fencing may be required to allow the recovery of riparian vegetation in

frequently heavily used areas. This may help to prevent the loss of important nest trees or critical Weidemeyer's admiral habitat. Temporary fencing can also be used to exclude cattle from critical northern leopard frog, plains spadefoot or Great Plains toad breeding ponds during the spring.

1.3 Burning or Mowing

Prescribed burning and mowing are other tools that have been suggested to mimic natural disturbance processes and maintain or enhance habitat for wildlife such as grassland birds. The application of these tools is contingent on further research as well as on buy-in and acceptance from landowners. Further research is needed to investigate historic fire return intervals and the potential impacts of mowing or burn treatments on plant community composition, structure and wildlife habitat. Mowing and burning are typically suggested as tools to control woody species encroachment or to reduce dense accumulations of litter. Their application is generally better suited to areas with higher moisture and greater grassland productivity. It is important to consider the timing, spatial extent and distribution and frequency of burning and mowing applications. To minimize their potential impacts to ground-nesting birds, mowing or burn treatments should not be conducted during the peak nesting season (*i.e.*, from May 1 to the end of July).

1.4 Summary Tables

The following tables are intended to provide an overview and quick reference to summarize the information presented in Part III of this report. Table S-1 provides a summary of habitat associations, key prey and timing considerations for the 33 species considered in this report. Tables S-2 and S-3 provide an overview of key BMPs for the Milk River and South Saskatchewan River watersheds.

Table S-1 Summary of Habitat Associations, Key Prey and Timing Considerations for 33 Select Management Species in the Milk River and South Saskatchewan River Watersheds

Species	Status in Alberta[1]	Habitat Type(s)	Key Features (B = Breeding Habitat; F= Foraging Habitat; W= Over-wintering Habitat)	Habitat Suitability Index Variable(s)[2]	Key Prey (if applicable)	Timing and Set-back Recommendations[3]
Birds						
Ferruginous hawk	At Risk	River valley, native prairie, riparian	B: Cliff and hoodoo nest sites along the Milk River and tributaries, tree nests, native prairie ground nests F: Native prairie with Richardson's ground squirrels	1. Native prairie 2. Moderately coarse soil texture	Richardson's ground squirrel	**Nesting sites:** Date: March 15 to July 15 Activity Level – Distance High – 1,000 m Medium – 1,000 m Low – 1,000 m Date: July 16 to March 14 Activity Level – Distance High – 1,000 m Medium – 100 m Low – 50 m
Swainson's hawk	Secure	Native prairie, tame pasture, riparian	B: Trees/shelterbelt nest sites, native prairie ground nests F: Native prairie with Richardson's ground squirrels and interspersed taller grasslands	N/A	Richardson's ground squirrel	**N/A** Breeding: March 15 to July 15
Golden eagle	Sensitive	River valley, native prairie	B: Cliff and hoodoo nest sites F: Shrubby, sagebrush or edge habitats	N/A	White-tailed jackrabbit, mountain cottontail	**Nesting sites:** Date: March 15 to July 15 Activity Level – Distance High – 1,000 m Medium – 1,000 m Low – 1,000 m Date: July 16 to March 14 Activity Level – Distance High – 1,000 m Medium – 100 m Low – 50 m

Short-eared owl	May Be At Risk	Native prairie, tame pasture	B: Ground nests in ungrazed/lightly grazed grasslands F: Meadow vole habitat (taller grasslands)	N/A	Meadow vole	**Active nest and surrounding habitat:**	
						Date: April 1 to July 15	
						Activity Level	Distance
						High	100 m
						Medium	100 m
						Low	100 m
Prairie falcon	Sensitive	River valley, native prairie	B: Cliff and hoodoo nest sites F: Native prairie with Richardson's ground squirrels within 15 km of nest sites	1. Slopes greater than 75° 2. Richardson's ground squirrel habitat suitability 3. Distance to ground squirrel habitat (15 km from nests)	Richardson's ground squirrel	**Nesting sites:**	
						Date: March 15 to July 15	
						Activity Level	Distance
						High	1,000 m
						Medium	1,000 m
						Low	1,000 m
						Date: July 16 to March 14	
						Activity Level	Distance
						High	1,000 m
						Medium	100 m
						Low	50 m
Sharp-tailed grouse	Sensitive	Native prairie, riparian	B: Uses native prairie with raised or flat areas of shorter cover for leks and nests in surrounding taller grass or shrub patches W: Aspen bluffs and shrubby or riparian hardwood draws	1. Native prairie 2. 5% to 15% shrub cover	N/A	**Leks:**	
						Date: March 15 to June 15	
						Activity Level	Distance
						High	500 m
						Medium	500 m
						Low	500 m
						Date: July 16 to March 14	
						Activity Level	Distance
						High	500 m
						Medium	100 m
						Low	100 m

Mountain plover	At Risk	Native prairie	B+F: Ground nests in short vegetation (less than 10 cm in height), between 30% and 50% bare ground and in areas with 0.5 km to 1 km diameter of flat terrain. Heavily grazed/ recently burned areas preferred.	N/A	N/A	**Active nest and surrounding habitat:** Date: April 1 to July 15 Activity Level – Distance High – 100 m Medium – 100 m Low – 100 m
Burrowing owl	At Risk	Native prairie	B: Uses Richardson's ground squirrel or American badger burrows with surrounding short vegetation for nesting F: Taller grassland cover (~30 cm to 60 cm) within 600 m of nests; wetland and riparian areas may be important	1. Native prairie 2. Moderate to moderately coarse soil texture 3. 0% shrub/tree cover 4. Distance to linear disturbance (>800m)	Deer mouse and meadow vole	**Nesting sites:** Date: April 1 to Aug. 15 Activity Level – Distance High – 500 m Medium – 500 m Low – 200 m Date: Aug. 16 to Oct. 15 Activity Level – Distance High – 500 m Medium – 200 m Low – 200 m Date: Oct. 16 to March 31 Activity Level – Distance High – 500 m Medium – 100 m Low – 50 m
Loggerhead shrike	Sensitive	Native prairie, tame pasture with shrubs or shelterbelts	B: Shrubs and shelterbelts used for nesting F: >20 cm tall grass cover, available tall shrubs or fences for perching and edge habitat	1. 5% to 15% shrub cover 2. >80% grass cover 3. Flat terrain 4. Farmyards	N/A	**N/A** Breeding: May 1 to July1

Long-billed curlew	Sensitive	Native prairie	B: Nests in short (10 cm to 20 cm) grasslands, in flat to rolling topography; taller vegetation is used for brood-rearing. Avoids areas with high shrubs. F: Short (10 cm to 20 cm) grasslands	N/A	N/A	**Active nest and surrounding habitat:** Date: April 1 to July 15 Activity Level / Distance High: 100 m Medium: 100 m Low: 100 m
Upland sandpiper	Sensitive	Native prairie, tame pasture	B: Nests in taller grassland cover (10 cm to 64 cm) and uses short to intermediate grasslands for brood-rearing (less than 15 cm). Prefers low to moderate forb cover and little woody vegetation cover. F: Shorter grass cover	N/A	N/A	**Active nest and surrounding habitat:** Date: April 1 to July 15 Activity Level / Distance High: 100 m Medium: 100 m Low: 100 m
Sprague's pipit	Sensitive	Native prairie	B+F: Native prairie; intermediate grass heights (~28 cm max), litter depths 1.2 cm – 3 cm, moderate to high grass to forb ratios, low shrub cover	1. Native prairie (>25%) 2. Less than 15% shrub cover 3. Distance from riparian areas	N/A	**Active nest and surrounding habitat:** Date: April 1 to July 15 Activity Level / Distance High: 100 m Medium: 100 m Low: 100 m
Brewer's sparrow	Sensitive	Native prairie, tame pasture	B: Native prairie with suitable shrub cover. In transition habitats between shrub-steppe and mixedgrass prairie, Brewer's sparrows prefer to nest in areas with lower sagebrush cover (5% to 10%), a denser understory of graminoids and forbs (64% to 73%) and shorter sagebrush plants (0.25 m to 1.0 m). F: Shrub patches	N/A	Insects	**N/A** Breeding: April 15 to September 15
Baird's sparrow	Sensitive	Native prairie, tame pasture	B+F: Native prairie or tame pasture; less than 20% shrub cover, litter depths of 0.1 cm to 4 cm and average grass heights of 10 cm to 30 cm	N/A	N/A	**N/A** Breeding: May 15 to July 15

Grasshopper sparrow	Sensitive	Native prairie, tame pasture, cropland	B: Native prairie with good grass cover and some shrubs F: Native prairie with more bare ground, fewer shrubs and lower litter cover compared to breeding habitat	N/A	Insects	**N/A** Breeding: May 1 to August 15
McCown's longspur	May Be At Risk	Native prairie, tame pasture, cropland	B: native prairie with lower cover of vegetation, litter and shrubs and higher cover of bare ground F: Unknown	N/A	Seeds, insects	**N/A** Breeding: April 1 to August 15
Chestnut-collared longspur	At Risk	Native prairie, tame pasture, cropland	B: Native prairie with less than 20 cm to 30 cm tall, minimal litter cover, scattered shrubs, and high bare soil cover.	N/A	Seeds, insects	**N/A** Breeding: April 1 to August 15
Mammals						
Olive-backed pocket mouse	Sensitive	Native prairie	B+F: Prairie with loose sandy soils (sandhills and sandy ecological sites), short or sparse vegetation, low shrub density	1. Grassland habitats 2. Moderate, moderately coarse or coarse soil texture 3. 10% to 30% bare ground 4. 60% to 80% graminoids 5. 5% to 10% shrub cover	N/A	**N/A** Breeding: late April to June Over-wintering: October 15 to April
Western harvest mouse	Undetermined	Native prairie	B+F: Dry mixedgrass prairie with good vegetation and litter cover	N/A	N/A	**N/A** Breeding: March to November (U.S.)
Swift fox	At Risk	Native prairie	B: American badger burrows, short vegetation, flat to rolling terrain F: Patchy vegetation (short – intermediate cover)	1. Flat terrain (<30°) 2. Silty soils 3. Low shrub cover (<5%) 4. High grass cover (>25%)	Voles are an important winter prey source	**Dens:** Date: Feb. 16 to July 31 Activity Level – Distance High – 500 m Medium – 500 m Low – 500 m Date: Aug. 1 to Feb. 15 Activity Level – Distance High – 500 m Medium – 100 m Low – 50 m

American badger	Sensitive	Native prairie, tame pasture	B: Open grasslands, friable soils, little woody cover F: Habitats with high concentration of Richardson's ground squirrels/northern pocket gophers	1. Moderately coarse, medium and moderately fine soil texture 2. Greater than 70% graminoid cover 3. 0° to 15° slopes 4. >400 m from roadways	Richardson's ground squirrel, northern pocket gopher	**N/A** Breeding: Birthing occurs from April to June
Richardson's ground squirrel	Secure	Native prairie, suitably grazed tame pasture	B+F: Short grassland vegetation (less than 5 cm) and moderately well drained soils	1. >20% grass cover 2. 0° to 10° slopes 3. Moderately coarse or medium soil texture	N/A	**N/A** B: March to April; juveniles emerge aboveground in May W: Hibernate for 4 to 8 months beginning in mid-June (adult males), July (adult females), or August to October (juveniles)
Mule deer	Secure	Native prairie, tame pasture, riparian	B: Dense shrub cover (*e.g.*, deciduous thickets) F: Rugged, open terrain, grass-forb-shrub interface W: South- and west-facing, wind-swept, grassy slopes along coulee breaks and slopes, shelterbelts, aspen groves	N/A	N/A	**N/A** B: Rutting: Late October to early December; Fawning: May to early June W: Winter range is used when snow depths exceed ~20 cm
White-tailed deer	Secure	Primarily riparian	B: Woody draws, drainages and basins with slopes of less than 15% F: Riparian habitats W: South- and west-facing, wind-swept, grassy slopes along river valleys, shelterbelts, aspen groves	N/A	N/A	**N/A** B: Rutting: Late October to early December; Fawning: May to early June W: Winter range is used when snow depths exceed ~20 cm

Reptiles and Amphibians						
Prairie rattlesnake	Sensitive	Native prairie	B: Breeds within 5 km of major drainages. South, east and southeast aspect are usually selected. Mammal burrows, flat rocks and moderate shrub cover are important habitat components. F: Native prairie within 25 km of major drainage W: Hibernacula found along river escarpments with stable slump blocks, meander scars and fissures, subterranean water channels, sinkholes and rocky outcrops. American badger burrows are often selected for. Hibernacula are usually located on south, east and southeast aspects.	Hibernacula (Wintering Habitat): 1. <4 km from major river, coulee or drainage 2. South, east, or southeast slopes 3. Rough terrain (high relief, moderate to steep slopes, >10% exposed bedrock) Summer Habitat: 1. <25 km from major river, coulee or drainage 2. Low densities of major roadways 3. Native prairie cover 4. Low to moderate shrub density Birthing Sites (Rookery): 1. 1 km to 5 km from major river, coulee and drainage 2. South, east, southeast aspect 3. Moderate shrub cover 4. Moderate bare rock cover	Generalist diet; small mammals important (voles, northern pocket gopher, Richardson's ground squirrel)	**Hibernacula:** Date: Year-round Activity Level / Distance High: 500 m Medium: 500 m Low: 200 m **Rookeries:** Date: Mar. 15 to Oct. 31 Activity Level / Distance High: 200 m Medium: 200 m Low: 200 m Date: Nov. 1 to Mar. 14 Activity Level / Distance High: 200 m Medium: 50 m Low: 50 m
Bullsnake	Sensitive	Native prairie	B: Sandy or friable soils; mammal burrows; south, east and southeast aspects along major river, coulee or drainage F: Native prairie within 25 km of major drainage; will forage in riparian forests W: Uses similar hibernacula as prairie rattlesnakes	N/A	Generalist diet; small mammals important (voles / northern pocket gopher / Richardson's ground squirrel)	**Hibernacula:** Date: Year-round Activity Level / Distance High: 500 m Medium: 500 m Low: 200 m

						Rookeries:
						Date: Mar. 15 to Oct. 31
						Activity Level / Distance
						High / 200 m
						Medium / 200 m
						Low / 200 m
						Date: Nov. 1 to Mar. 14
						Activity Level / Distance
						High / 200 m
						Medium / 50 m
						Low / 50 m
Plains hognose snake	May Be At Risk	Native prairie	B+F: Native prairie with sandy surficial deposits, sufficient vegetation cover and with seasonal or permanent wetlands W: Rodent burrows	N/A	Toads	**Hibernacula:**
						Date: Year-round
						Activity Level / Distance
						High / 500 m
						Medium / 500 m
						Low / 200 m
						Rookeries:
						Date: Mar. 15 to Oct. 31
						Activity Level / Distance
						High / 200 m
						Medium / 200 m
						Low / 200 m
						Date: Nov. 1 to Mar. 14
						Activity Level / Distance
						High / 200 m
						Medium / 50 m
						Low / 50 m

Short-horned lizard	At Risk	Native prairie	B+F: Sparsely vegetated south-facing slopes of river valleys and coulee preferred; some sagebrush cover and mammal burrows provide thermal cover W: Loose soil of south-facing slopes	1. <100 m of valleys 2. Native prairie 3. <1,200 m elevation 4. <5% riparian features	Ants	**Habitat:** Date: Year-round Activity Level / Distance High: 200 m Medium: 100 m Low: 100 m
Eastern yellow-bellied racer	Accidental / Vagrant	Native prairie	B: In open dry mixedgrass prairie along river valleys F: Lowland pastures, mudflats and riparian areas. Prefers micro-habitats with higher shrub cover, higher vegetation density, and greater proximity to burrows. W: Deep holes in soft soil on hillside slopes; mammal burrows; deep crevices in or near limestone rock ledges; hillside slumps; and abandoned cisterns.	N/A	Insects, small rodents, lizards, amphibians	**N/A** Breeding: spring
Great Plains toad	Sensitive	Native prairie	B: Ephemeral wetlands, fresh, clear water, in areas of sandy soil F: Native prairie near ephemeral wetlands with sandy, friable soils W: Rodent burrows, friable sandy soils	1. >75% native prairie 2. Chernozemic / Solonetzic soil orders 3. Moderately coarse and coarse soil texture	Insects	**Breeding ponds:** Date: Year-round Activity Level / Distance High: 100 m Medium: 100 m Low: 100 m
Canadian toad	May Be At Risk	Native prairie	B: Seasonal and permanent wetlands, margins of lakes F: Wetlands or adjacent upland areas W: Upland areas with sandy soils	N/A	Insects	**N/A** Breeding: May
Plains spadefoot toad	May Be At Risk	Riparian	B: Ephemeral wetlands F: Native prairie with sandy soils; ephemeral wetlands W: Friable sandy soils, rodent burrows	1. Moderately coarse and coarse soil texture 2. >75% native prairie	Insects	**Seasonal wetlands in native prairie:** Date: Year-round Activity Level / Distance High: 100 m Medium: 100 m Low: 100 m

Northern leopard frog	At Risk	Riparian	B: Shallow, warm standing water of permanent waterbodies with abundant aquatic and emergent vegetation F: Near waterbodies; open areas with intermediate vegetation (avoids heavily grazed and tall, dense vegetation) W: Springs, deep waterbodies that do not freeze solid	N/A	Insects	**Breeding ponds:**	
						Date: Year-round	
						Activity Level	Distance
						High	100 m
						Medium	100 m
						Low	100 m
Other							
Weidemeyer's admiral butterfly	May Be At Risk	Native prairie, riparian	B+F: Shrub complexes and woody riparian vegetation along the valleys of the Milk River and its tributaries, including spring or seepage sites and coulee valleys	1. Valley topography 2. >10% shrub cover	N/A	**N/A** Flight period: mid-June to mid-July	

[1] Source: Government of Alberta (GoA) (2016)
[2] Source: Downey *et al.* (2004)
[3] Source: GoA (2011)

Table S-2 Summary of Beneficial Management Practices for the Milk River and South Saskatchewan River Watersheds

GENERAL LAND-USE RECOMMENDATIONS
Native Habitat Protection
▪ Protect large, contiguous blocks of native prairie from cultivation or extensive development. It is preferable to maintain native grassland habitats with high interior area and minimal edge to reduce potential edge and isolation effects (Johnson and Igl 2001). Grassland bird studies have shown that nest predation and nest parasitism rates are higher in small isolated habitat patches or in edge habitats (Johnson and Temple 1990). ▪ Limit disturbance to unique or rare habitat types such as sand hills/badlands/hoodoos and cliffs along major river valleys such as the North Milk and Milk Rivers. ▪ Minimize disturbance to sparsely vegetated south, southeast or east facing slopes and slump areas along river valleys and coulees and other major drainages at any time of the year. These areas provide critical overwintering habitat to short-horned lizards, prairie rattlesnakes and bullsnakes. ▪ Maintain or establish natural corridors between existing native prairie habitats or natural landscape features such as wetlands. ▪ Minimize the spread of exotic plants (*e.g.*, smooth brome [*Bromus inermis* ssp. *inermis*], timothy [*Phleum pratense*]) in native habitats. The tall or dense growth habits of many herbaceous invasive plants can diminish habitat quality for several wildlife species, including mountain plovers, Sprague's pipits, Baird's sparrows, swift fox, Richardson's ground squirrels, prairie rattlesnakes, bullsnakes, plains hognose snakes, short-horned lizards, plains spadefoots, Great Plains toads and northern leopard frogs. ▪ Retain shrubby and woody draw mosaics across the landscape. Shrub patches provide important habitat for Brewer's sparrow, loggerhead shrikes, sharp-tailed grouse, white-tailed and mule deer, and for important raptor prey species such as mountain cottontails. Shrubby areas provide these species with breeding, shelter, or foraging habitat. Shrub complexes and woody riparian vegetation along the Milk River and its tributaries are considered critical habitat for Weidemeyer's admiral butterflies and provide important foraging habitat for western small-footed bats. ▪ Where the goal is to maintain habitat for raptors, maintain individual trees or interspersed tree stands in the landscape and leave dead or decadent trees standing. Monitor the condition of nest trees and where necessary, plant trees or reinforce dead or decadent trees. ▪ Reclaim and revegetate heavily impacted riparian corridors with native vegetation. ▪ Where possible, convert non-native uplands (*e.g.*, marginal cropland) to native vegetation or permanent cover. ▪ Maintain and restore the health of wetland, stream and riverine riparian habitats. Restore formerly drained wetlands and avoid draining wetlands or converting ephemeral wetlands into permanent waterbodies. ▪ Maintain vegetated buffers around wetlands.

GENERAL LAND-USE RECOMMENDATIONS

Pest Control / Farming Practices

- Promote organic farming practices.
- Consider planting winter wheat as a crop alternative, where feasible. Winter wheat fields require less disturbance during the spring and early summer and thereby minimize possible disturbance to ground nesting birds during the breeding season.
- Promote no-till or minimal tillage and minimize plowing during the spring and early summer nesting season. Zero tillage is beneficial to ground nesting birds.
- Delay haying or harvesting crops until late July or August after the peak nesting period. Use flushing bars during haying to minimize mortality of ground nesting birds.
- Retain shelterbelts around old farmsteads and field edges to provide nesting or cover habitat for raptors, loggerhead shrikes and deer. Consider diversifying shelterbelts by planting native thorny shrubs such as thorny buffaloberry (*Shepherdia argentea*) or hawthorn (*Crataegus* spp.). Where practical, plant multiple, irregular patches of shrubs or trees along shelterbelts to create larger blocks of habitat and reduce the linear nature of shelterbelts.
- Minimize synthetic fertilizer use near drainages and wetlands.
- Avoid the use of organochlorine pesticides including carbamates and organophosphates. These pesticides reduce the diversity and abundance of food sources for wildlife, and may be directly or indirectly toxic or negatively impact reproductive success.
- Discontinue Richardson's ground squirrel poisoning control programs. Richardson's ground squirrel populations are an important prey source for ferruginous hawks, Swainson's hawks, prairie falcons, golden eagles, American badgers, swift fox, rattlesnakes, bullsnakes and plains hognose snakes. Their burrows also provide important shelter or breeding habitat for burrowing owls, mountain plovers, olive-backed pocket mice, rattlesnakes, bullsnakes, hognose snakes and spadefoot and Great Plains toads.
- Promote and maintain viable populations of natural predators such as raptors and badgers to provide an alternate method of rodent control.

GENERAL LAND-USE RECOMMENDATIONS
Industrial / Commercial Development / Human Disturbance
▪ Minimize human disturbance near critical wildlife habitats (leks/nests/dens/breeding ponds/ hibernacula/winter range). ▪ Abide by Alberta Environment and Parks set-back and timing restrictions for land-use activities or developments near critical wildlife habitats. Constructing during the winter is preferable for most species to avoid disturbance during sensitive breeding periods, provided key winter range or hibernacula are not disturbed. ▪ Conduct pre-development wildlife surveys and plan developments to avoid sensitive or unique natural habitats where possible. ▪ Use the Milk River Watershed Habitat Suitability Index (HSI) maps and models as a planning tool to avoid sensitive areas and to schedule wildlife surveys. Check for HSI model revisions and updates. Update HSI models using Resource Selection Functions wherever possible. ▪ Use appropriate construction methods to minimize development impacts and facilitate reclamation of native habitats. ▪ Avoid leaving open trenches during the spring and summer months. During construction, routinely check and remove any snakes, amphibians or other wildlife species from trenches. ▪ Avoid development of water diversion projects such as creation of new dams or reservoirs along prairie rivers and drainages in the Milk River and South Saskatchewan River watersheds. Altered hydrology impacts floodplain and wetland habitats with negative consequences to numerous species. Localized flooding at dam or reservoir sites can eliminate critical habitat for rare species.

Table S-3 Summary of Beneficial Grazing Practices for the Milk River and South Saskatchewan River Watersheds

GRAZING RECOMMENDATIONS
General Goals
▪ Avoid uniform, heavy grazing across large areas for prolonged periods. Promote patchy grazing to create heterogeneous habitat patches with variable canopy cover, litter (organic residue) and plant species diversity. A mosaic of cover types creates habitat for a wider range of flora and fauna. The desired size or distribution of light, moderate or heavily grazed patches depends on the suite of wildlife species being managed for. ▪ Manage and monitor cattle use of upland woody vegetation to ensure trees and shrubs are healthy and capable of regenerating. ▪ Consider local ecological conditions, the distribution of valued habitat components, and the life histories of priority and select management species when developing grazing management plans. ▪ Pay attention to appropriate management of cattle in riparian areas. ▪ Apply a flexible and adaptive approach to grazing management. ▪ Monitor range and riparian health and adjust stocking rates in relation to health scores as well as wildlife habitat values. Consider the contribution of wild grazers when setting stocking rates. Reduce stocking rates during drought periods. Proper use factors of 25% to 50% are recommended for native prairie. ▪ Promote the use of multiple grazing systems best suited to meet local conditions and management goals of local operations. Multiple grazing systems will increase the range of variability on a landscape scale. ▪ Promote rest periods to restore cover values for wildlife species. For example, leaving one field or a portion of a field undisturbed during the peak nesting period (from May to mid-July) may increase cover available to ground nesting birds. ▪ Carefully plan the development of new livestock facilities (*e.g.*, watering facilities, fences, corrals, handling facilities, *etc.*) in relation to critical habitat features such as leks, hibernacula, traditional nesting sites or key winter ranges. ▪ Minimize the use of supplemental feeding practices that allow for the potential spread of exotic species. Where possible, limit feeding stations to existing cultivated areas or tame pastures and avoid placing feeding stations near to critical wildlife habitat features. ▪ Avoid creating dugouts in ephemeral or permanent wetlands to maintain their value as breeding sites for amphibians. ▪ Defer early season use of native prairie where possible, but avoid converting native prairie to tame pasture to accommodate deferred use. ▪ Use livestock distribution tactics (*e.g.*, salt or water) to minimize impact to sensitive wildlife habitats, create heterogeneous cover, or to enhance habitat for species like the mountain plover which benefits from heavier use.

GRAZING RECOMMENDATIONS
Riparian Area Management
▪ Careful management of riparian areas can help to retain their value as important habitats to numerous wildlife species. Riparian habitats associated with streams, rivers and wetlands in the Milk River and South Saskatchewan River watersheds provide important habitat for northern leopard frogs, Great Plains toads, sharp-tailed grouse, loggerhead shrikes, ferruginous hawks, golden eagles, Swainson's hawks, Weidemeyer's admirals, mule deer and white-tailed deer. ▪ Defer use of riparian areas during vulnerable periods (*i.e.*, spring and fall) when banks are saturated and easily damaged or when woody species are vulnerable to browsing. ▪ Observe appropriate utilization rates. ▪ Use off-stream watering and salt placement to reduce impact to riparian areas. ▪ Provide gravelled or hardened access points for livestock along rivers or streams at high use areas. ▪ Promote the use of deferred or rest-rotation grazing systems, where appropriate, to provide periods of rest and recovery. ▪ Consider the use of riparian pastures in areas with distinct upland and lowland areas to create more homogenous vegetation units and better control livestock distribution. Riparian pastures are created by fencing off riparian areas as separate units from the upland. ▪ Consider the use of temporary exclusion fencing to improve severely degraded riparian areas or protect critical habitats such as northern leopard frog breeding ponds.

GRAZING RECOMMENDATIONS

Grazing System Properties – Comparative Summary

Continuous (Season-Long) Grazing

Properties:

- Livestock are held within one grazing unit (field) for the duration of the grazing season (usually the active growing season).

Potential impacts:

- Can create patchy grazing under appropriate light to moderate stocking rates and with effective use of livestock distribution techniques. Patchy grazing is the result of selective grazing where forage supply exceeds demand. Grazed patches are maintained by repeated grazing of regrowth that is preferred to more mature vegetation. Patchy grazing creates heterogeneous grassland and more diverse habitat for wildlife.
- Has the potential to heavily impact riparian areas and associated wildlife habitat. Potential for impact to riparian areas is increased under heavy stocking rates, limited water sources, ineffective use of livestock distribution techniques, and in pastures with variable terrain and distinct upland and lowland areas.
- Potential impacts to breeding areas depend on livestock entry dates. Generally, season-long grazing does not defer use during the early season and, therefore, does not allow for control of potential impacts to breeding areas or other critical wildlife habitats during the spring.
- Temporary fencing may be required to prevent impact to sensitive features at critical times of the year under this grazing system.
- Season-long grazing during the fall and winter is considered appropriate for fescue prairie. Winter or late season use in fescue prairie may benefit wildlife such as grassland birds by reducing litter build-up or creating favourable vegetation structure, with minimal impacts during the breeding season. Fall grazing of fescue prairie can also improve the range for mule and white-tailed deer by removing old growth and stimulating the production of new growth. This can improve the quality and availability of spring forage for deer.

Switchback / Deferred / Rest-rotation

Properties:

- Deferred early season grazing is rotated among two fields (switchback), or among three or more fields (deferred rotation grazing). Under deferred rotation grazing fields grazed first in one year (early) are grazed last (late) during the next year and grazed second (mid) in the subsequent year. Rest-rotation grazing requires a minimum of four fields to be implemented, with one field rested from grazing for an entire year, and other fields grazed either early, mid or late in the year. Grazing use or rest is rotated between fields in a sequential fashion.

GRAZING RECOMMENDATIONS

Potential impacts:

- Allows a field or part of a field to be rested during critical periods to wildlife such as the spring breeding season.
- Early-season deferral may improve cover values for ground nesting birds such as sharp-tailed grouse, short-eared owls, Sprague's pipits and Baird's sparrows. Spring deferral helps lessen disturbance to these birds during breeding, nesting and peak hatching. Deferred early season use can also be used to limits impacts to leopard frog, plains spadefoot toad and Great Plains toad breeding ponds and reptile hibernacula and birthing areas.
- Allows for deferred use in riparian areas during sensitive periods and permits periodic rest of these areas.

Complementary Grazing

Properties:

- Native prairie is fenced out as a separate grazing unit from tame pasture. Tame pastures are grazed in the spring (from late April to mid-June) and in the fall (mid-September to October).

Potential impacts:

- Allows for deferred early season use of native prairie, minimizing potential impacts during sensitive breeding periods.
- This system is most beneficial to wildlife if tame pastures already exist within the grazing operation or where it is possible to convert marginal cropland to tame pasture. Benefits to wildlife are diminished where tame pasture is created at the expense of native prairie.
- Tame pasture provides supplemental forage for mule and white-tailed deer in the spring and fall.

Intensive Grazing (e.g., high-intensity-low frequency grazing/short duration grazing)

Properties:

- Intensive grazing systems involve very high stocking rates and utilization followed by long periods of rest. Under intensive grazing systems there is rigid control of animal distribution with the use of numerous smaller grazing units. Continual monitoring is required to adjust grazing periods and stocking rates and to match the prevailing growing conditions.

Potential impacts:

- Heavy stocking rates with repeated high intensity trampling can reduce soil infiltration rates and increase erosion and lead to declines in range condition, lower root mass and

GRAZING RECOMMENDATIONS
lower vegetation densities. ▪ Heavy stocking rates, even for short periods can have negative impacts on critical habitat features such as breeding ponds or riparian habitats. ▪ Intensive systems can create uniform grazing effects and diminish nesting, escape or thermal cover for ground nesting birds (*e.g.*, sharp-tailed grouse, Sprague's pipit and Baird's sparrow), small mammal prey species (*e.g.*, meadow voles), and reptiles (*e.g.*, prairie rattlesnake and plains hognose snakes). Potential impacts to wildlife depend on the duration of intervening rest periods and the timing of grazing. ▪ Intensive grazing may create suitable habitat for mountain plovers and Richardson's ground squirrels and may improve foraging opportunities for raptors such as ferruginous hawks, Swainson's hawks and golden eagles.

1.5 Closing Remarks

The recommendations provided in this report provide a general framework for multi-species habitat conservation as part of the MULTISAR project. However, these recommendations are not exhaustive or static. Ongoing research initiatives will help to provide management direction and set management priorities. Recommendations should be reviewed, added to and amended where necessary with feedback from appropriate agencies, landowners and other interested parties.

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