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# Home ranges, resource selection, and parasite diversity of urban versus rural elk (*Cervus elaphus*)

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UNIVERSITY OF CALGARY

Home ranges, resource selection, and parasite diversity  
of urban versus rural elk (*Cervus elaphus*)

by

Benn Craig Edwards

A THESIS

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## **Abstract**

Increased human development and accessibility to nature preserves is a mounting pressure on wild populations, forcing animals to abandon areas or modify behavior in response to habitat fragmentation. In this study, I examined the effects of development within the Bow Valley corridor on: home ranges, resource selection, and parasites; in four populations of urban and rural elk (*Cervus elaphus*). Total home range and core home range size was smaller in urban herds (Canmore and Banff). In proportion to habitat availability, urban animals selected against steep slopes, high elevations, cover habitat (white spruce), and high road density, while selecting for grazing habitat. Rural animals (Bow Valley Provincial Park and Deadman's Flats) selected against steep slopes and high road densities and in favor of grazing habitat types, but selected for cover habitat. Overall, parasite prevalence, intensity, and diversity were greatest in the urban centers and lowest in rural herds.

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### List of Symbols and Abbreviations

$\alpha$ -LoCoH	Adaptive Local Convex Hull (Getz 2007)
AVI	Alberta Vegetation Inventory
BCEAG	Bow Corridor Ecosystem Advisory Group
CCOG	Canada Council on Geomatics
CDED	Canadian Digital Elevation Data
DEM	Digital Elevation Model
ESRI	Environmental Systems Research Institute (Redlands, CA)
FEC	Fecal Egg Count
GEE	Generalized Estimating Equation
GIS	Geographic Information System
GLM	Generalized Linear Model
GLMM	Generalized Linear Mixed Model
GME	Geospatial Modeling Environment (Hawthorne Beyer, 2012)
GPS	Geographic Positioning System
IT	Information Theoretic (approach to model selection)
LCP	Least Cost Path Analysis (for corridor or path selection)
NHN	National Hydro Network
NRN	National Road Network
MCP	Minimum Convex Polygon
MSHC	Minimum Spurious Hole Covering
PROC GENMOD	Generalized Linear Model Procedure in SAS Base 9.3
QIC	Quasi-Likelihood under the Independence Model Criterion
QP 3.0	Quantitative Parasitology 3.0 (Reiczigel, J. & Rosza, L. 2005)
\ RSF	Resource Selection Function
RSPF	Resource Selection Probability Function
SAS	SAS 9.3 TS Level 1M0 (SAS Institute Inc. 2010)

## **1 Introduction**

### **1.1 Habitat Fragmentation**

Increased human development and accessibility to nature reserves is a mounting pressure on wild populations, forcing animals to abandon areas or modify behavior in response to habitat fragmentation and anthropogenic influences such as buildings and transportation corridors (Theobald, Miller et al. 1997; Manor and Saltz 2003). Animals surviving in habitat fragments are isolated in a reduced landscape with novel ecological boundaries, and must cope with these disturbances; sometimes negatively impacting patterns of habitat-use, foraging efficiency, survival, and fitness (Manor and Saltz 2003; Ewers and Didham 2006). Life history of animals exploiting fragmented versus un-fragmented habitats will often differ, and the adaptability of social and behavioral strategies is often a determining factor in the survival of disrupted populations (Banks, Piggott et al. 2007).

Habitat fragmentation involves an alteration of the spatial distribution and quality of resources available to animals, as well as a reduction of total area and increased dispersion of suitable habitat (Banks, Piggott et al. 2007). Associated changes in vegetation community structure and human disturbance through recreational or industrial activity can further reduce resource availability both within patches, as well as cumulatively across landscapes (Ims, Rolstad et al. 1993).

Ungulates displaced by human disturbance typically increase evasive behavior. This response ranges from short-term flight to area abandonment and shifts in home range (Manor and Saltz 2003; Manor, Saltz et al. 2005). This increase in vigilance can come at the expense of foraging behavior or reproductive success (Phillips and Alldredge 2000; Manor, Saltz et

al. 2005). For example, elk (*Cervus elaphus*) disturbed by humans during calving have demonstrated a significant decline in reproductive success (Phillips and Alldredge 2000), while mountain gazelle (*Gazella gazella*) display avoidance of altered habitats and exhibit greater flight distance with increased human presence (Manor, Saltz et al. 2005). On a molecular level, habitat fragmentation causes reduced genetic diversity in bighorn sheep (*Ovis canadensis*) due to losses in connectivity and gene flow among populations (Epps, Palsbell et al. 2005).

Despite the progression of human disturbance on a global scale, there are still comparatively few studies examining the disruption of biological processes in fragmented ecosystems, and our understanding of how processes will be altered and which taxonomic groups will be most affected is still poor; thus, the behavioral consequences of human disturbance merits further investigation (Gillespie and Chapman 2006; Banks, Piggott et al. 2007).

## **1.2 Effects of Wildlife Urbanization**

While many animals display avoidance in response to human development, those that tolerate change and adapt their foraging behaviors and habitat selection may continue to use these disrupted habitats and experience greater fitness (Rubin, Boyce et al. 2002). The tolerance to human presence can occur when disruption is both common and predictable, and interactions are not accompanied by negative reinforcement (Thompson and Henderson 1998; Manor and Saltz 2003). This behavioral strategy is predicted to maximize fitness as urban environments often present advantages as both predation refuges and a source of high-quality forage such as cultivated or landscaped vegetation (Thompson and Henderson 1998; Rubin, Boyce et al. 2002). Large mammalian predators are typically highly sensitive to



anthropogenic disturbances due to their large home ranges and foraging strategies, and tend to avoid areas affected by human recreation and development; thus creating refugia for prey species that are able to exploit these altered habitats (Van Dyke, Brocke et al. 1986; George and Crooks 2006). The exploitation of urban refugia may provide optimal habitat and foraging opportunities; for example, populations of bighorn sheep inhabiting urban areas have been found to have higher diet quality, greater reproductive success, and conserve energy by foraging on low elevation ranges with gentler slopes (Rubin, Boyce et al. 2002).

While the term “urban” is traditionally reserved for major metropolitan centres, within the context of this study I have appropriated it to describe the residential and commercial districts comprised within the towns (Canmore & Banff) and hamlets (Harvie Heights, Exshaw, Deadman’s Flats) of the Bow Valley. North American elk in the Bow Valley have exploited these “urban” refugia on a non-migratory basis for several decades and this pattern has begun to perpetuate in some of the wild ranging elk of Ya Ha Tinda Ranch (Hebblewhite, Merrill et al. 2006) and Rocky Mountain National Park (Lubow, Singer et al. 2002), among others (Thompson and Henderson 1998). Despite the immediate benefits of these altered ranging and foraging behaviors, the long-term implications on both species persistence and overall biodiversity are relatively unknown (Rubin, Boyce et al. 2002).

### **1.3 Habitat Selection**

All organisms are subject to the competing demands of survival and reproduction which encompass a multitude of behaviors including: foraging, mate selection, rearing of young, predator evasion and defense of limited resources (Bjorneraas, Herfindal et al. 2012). In order to maximize fitness, animals must adjust their location in space in order

to exploit the heterogeneously distributed resources which influence these objectives (Hebblewhite and Merrill 2009). Understanding the patterns of these dynamic spatial associations is a key component of unraveling the mechanisms which determine allocation of resources to the multitude of competing life-history demands and trade-offs which ultimately determine individual fitness (Johnson 1980 ; Beyer, Haydon et al. 2010).

Habitat or resource availability differs greatly across the range of spatial scales, but the spectrum of behavioral processes that occur in the course of selective behavior fall generally into four classes: (1) the entirety of a geographic distribution of a species, (2) establishment of a home range within that geographic expanse (can be herd level home range in gregarious species), (3) habitat use within the home range, and (4) fine-scale and temporally focused individual behavior, such as selection of particular food items within a patch (Johnson 1980 ; Beyer, Haydon et al. 2010). Thus, accessibility to resources and habitat-related ecological and behavioral processes are highly subject to the spatio-temporal scale at which selection is examined (Borger, Franconi et al. 2006).

In addition to effects of scale, intraspecific differences in life-history requirements can result in differences in resource selection between the sexes of dimorphic species (Bjorneraas, Herfindal et al. 2012). In gregarious, sexually dimorphic species; such as ungulates, mixed-sex groups are uncommon due to differences in activity budgets between the sexes (time spent foraging versus resting and selection of forage quality). These differences increase the cost of synchrony, which is detrimental to the maintenance of group cohesion (Ruckstuhl 1998; Ruckstuhl and Neuhaus 2002). These differences in activity budgets are also present within sexes depending on age, maturity, or lactation

status in females (Ruckstuhl 1998). As a result of social sexual segregation, resource availability may differ between the sexes and corresponding differences in habitat selection can occur (Bjorneraas, Herfindal et al. 2012) which makes patterns of habitat selection context-specific to both sex and demographics. I will explore female elk resource selection in Chapter 3 by contrasting resource selection of urban to rural elk, and discussing potential benefits and costs to living in an anthropogenically-altered landscape. Host parasite assemblages in shrinking ecosystems

Factors such as climate change (Kutz, Hoberg et al. 2004), habitat fragmentation (Puttker, Meyer-Lucht et al. 2008; Mbori and McPeck 2009), and anthropogenic habitat disturbance (Patz, Olson et al. 2008) alter ecosystems and species distributions. This often concentrates both individuals and species within restricted or spatially shifted ranges, which promotes the transmission of parasites. The contributing effects of environmental contaminants and habitat destruction may lead to immune suppression in these challenged animals (Holmes 1996). As parasite pathogenicity is often directly correlated with parasite numbers (Dobson and May 1986), factors that increase close contact and promote transmission (such as high population densities), are also likely to increase the degree of pathogenicity and the magnitude of parasite-related health effects on host populations (Holmes 1996; Hoberg, Kocan et al. 2008). In directly transmitted parasites, host density is considered to be the greatest parameter in determining infection rates; showing strong positive correlation with parasite prevalence and diversity (Gillespie and Chapman 2008). When pathogens achieve these high densities within reservoir populations spillover into secondary or aberrant hosts may also occur (Power and Mitchell 2004). This spillover has the potential to shape population and community

structure; in particular, the transmission of parasites originating from domesticated animals, which could have drastic effects on wild populations (Holmes 1996).

Host-parasite assemblages can be sensitive indicators of ecological change following human-mediated transformations such as climate change or habitat disruption (Hoberg, Polley et al. 2008; Kutz, Jenkins et al. 2009). These transformations may result in altered ranging, shifting inter/intraspecific associations, altered transmission of endemic parasites, host switching (Hoberg, Kutz et al. 2002), disease emergence (Dobson, Kutz et al. 2003; Davidson, Simard et al. 2011) or land-use changes (Kutz, Jenkins et al. 2009), all of which affect the host/parasite dynamic. In order for such interactions to serve as an indicators, baseline investigations on host-parasite assemblages must occur (Hoberg, Polley et al. 2008). A detailed survey of parasite fauna is a critical component to recognizing altered host-parasite dynamics, emergence of new pathogens, and interactions between domestic and sylvatic host populations and should be an integral component in the assessment of biodiversity at local, regional, or global scales (Brooks and Hoberg 2000; Hoberg, Kocan et al. 2008). In Chapter 4, I will be discussing the association between urbanization and parasite abundance/diversity in rural (or, wild land) elk versus “urbanized” individuals.

#### **1.4 North American elk**

North American elk (*Cervus elaphus*) are a large, sexually dimorphic, gregarious ungulate with a current natural distribution encompassing the provinces west from Manitoba, as well as the Rocky Mountain and West Coast states (Smits 1991; Toweill and Thomas 2002). Through the spring and summer months elk may be solitary;

particularly during calving and weaning periods, or congregate in small single-sex groups (Struhsaker 1967; Toweill and Thomas 2002). Male elk begin extensive rutting behavior in the early fall by guarding and herding females into harems in order to monopolize mating opportunities. Harem sizes can range from single digits to several dozen receptive females, in the case of locally dominant bulls (Toweill and Thomas 2002; Wolff and Horn 2003). Elk will typically maintain this large-group social structure through the winter in response to predation pressures. Group size as well as the ratio of vigilance to foraging behavior is highly dependent on the density of local predators, as well as prey density and group composition (Hebblewhite and Pletscher 2002; Toweill and Thomas 2002; Wolff and Horn 2003; Christianson and Creel 2008).

Elk are often considered a “keystone” species, important in ecosystem maintenance through their migrations, foraging, and essential role as prey in supporting large carnivore populations (Hebblewhite 2000; Hebblewhite, Merrill et al. 2006). As inhabitants of montane, foothill, and prairie ecosystems elk are often co-habitant with a variety of human developments including: recreational preserves, farming and ranching, residential expansions, and transportation corridors (Toweill and Thomas 2002).

Graminoids make up the bulk of elk diet which is also supplemented through browse - particularly coniferous sources over the winter and fresh aspen shoots in early spring (White, Feller et al. 2003; Baker, Ducharme et al. 2005; Christianson and Creel 2007). Foraging behavior is highly dependent on vegetation availability, and the ratio of grazing to browsing type feeding can be influenced by landscape composition. Elk will typically show preference for graminoids but when grazing patches are sparse or predation pressure is high, the animals will shift preference to browsing in deciduous or

coniferous stands (Christianson and Creel 2007; Christianson and Creel 2008). In high-density populations, intensive elk herbivory has the potential to result in the depletion of graminoid cover, increased erosion rates within feeding grounds and movement corridors, as well as reductions in biomass of willow and aspen stands (Hudson and Nietfeld 1985; Toweill and Thomas 2002; White, Feller et al. 2003; Baker, Ducharme et al. 2005).

Annual migratory behavior is not consistent for all populations of elk. While some animals migrate between cooler high-productivity alpine ranges in summer to lower elevation ranges in winter in response to increasing snow cover and deteriorating alpine forage conditions, several herds have been found to remain in winter ranges year-round (e.g. part of the Ya Ha Tinda Herd (Hebblewhite, Merrill et al. 2006) and Chamberlain Creek (Edge, Marcum et al. 1986)) when adequate supplies of forage, water, and cover are available (Toweill and Thomas 2002). Elk are considered to be a highly adaptable species and are able to tolerate considerable exposure to human developments (Toweill and Thomas 2002). This tolerance of humans and adaptability to exploit abundant resources on a non-migratory basis; combined with the encroachment of human development of high-quality low-elevation winter ranges, has the potential to accelerate elk habituation to human presence (Thompson and Henderson 1998; Haggerty and Travis 2006).

The Bow Valley elk herds provide an excellent system in which to examine how shrinking home ranges and changing availability of traditional and novel (*i.e.* anthropogenically modified) habitat can have specific consequences on resource selection, and result in the alteration of host-parasite assemblages.

Chapter 2 will be the theoretical basis for home range and resource selection models and will set the stage for Chapter 3 (Home Ranges and Resource Selection Functions of Bow Valle elk).

## 2 Technical Review: Home Range Estimates & Resource Selection Functions

### 2.1 Home Range Estimates

When defining how animals use a particular habitat, the most common descriptor has typically been home range size (Borger, Franconi et al. 2006; Borger, Franconi et al. 2006a). This parameter is superficially simple; however, conceptual differences in estimating the area have resulted in the creation of several parametric and non-parametric methods of home range delineation (Gitzen and Millsbaugh 2003; Borger, Franconi et al. 2006a; Kie, Matthiopoulos et al. 2010). The high resolution of modern GPS telemetry data has fueled a progression from “biased and inefficient” minimum convex polygons (MCP) (Borger, Franconi et al. 2006a; Kie, Matthiopoulos et al. 2010); which estimate home ranges as an area encompassed within the boundaries of the extreme satellite points within a telemetry dataset, to more robust kernel methods which are less sensitive to these outliers, and generate utilization distributions to describe habitat use within the home range (Kie, Matthiopoulos et al. 2010).

Adaptive local convex hull modeling ( $\alpha$ -LoCoH – Getz *et al.* 2007) is a recent approach to constructing home ranges and utilization distributions which has a greater success rate when detecting hard boundaries (*i.e.* impassable to wildlife) than typical kernel methods (Kernel Density Hulls) or MCPs (Getz and Wilmers 2004). The local convex hull method takes a nearest neighbor approach to hull (*i.e.* polygon) construction by establishing vectors between each GPS fix (as a focal point), and  $x$ -number of its nearest neighbors such that the sum of the length of these vectors equals a specific value,  $a_1$ , and once these vectors are established the area encompassed by the involved points is merged to construct a single hull (Getz, Fortmann-Roe et al. 2007). The use of a limiting



sum distance; rather than a set number of nearest neighbors (k-LoCoH) or set radius (r-LoCoH) around each focal point, results in an adjustment of the radius of the sphere around each focal point in such a way that smaller hulls are created in high use areas and create higher resolution data where points are abundant. This process of hull construction occurs iteratively until each telemetry point is involved as a focal point with its own unique hull, with the final union of these hulls being indicative of the home range of the animal from which the telemetry points were collected.

A utilization distribution may be constructed by sorting the hulls from smallest to largest and taking a union (*i.e.* merging) of the hulls until a specific percentage ( $x\%$ ) of all points are included thus creating the  $x\%$  isopleths in the utilization distribution (Getz, Fortmann-Roe et al. 2007). In this manner it is possible to define intensity of habitat use by isopleth level, and identify important or “core” areas of an individual or social group’s home range. Convention dictates that the entirety of an animal’s home range be defined as the 95% isopleth of the utilization distribution (Getz, Fortmann-Roe et al. 2007), although recent investigations have attempted to reduce home range over-estimates due to the inclusion of outlier points. By reducing the cutoff for total home range to values between the 50<sup>th</sup> and 90<sup>th</sup> percentile isopleths, the inclusion of spurious animal movements or exploratory behavior outside the boundaries of the true home range is reduced (Borger, Franconi et al. 2006a). However, this suggestion is largely in response to the spatial error inherent to conventional radiotelemetry triangulation methods - a source of uncertainty which is largely absent from GPS-based designs (Getz, Fortmann-Roe et al. 2007; Frair, Fieberg et al. 2010).

## 2.2 Resource Selection Functions

### Theory & Design

In this manuscript, I describe the use of resource selection functions for the investigation of habitat use in “urban” and rural elk (Chapter 3). In this section I will examine the limitations and advantages of inferring selection by using this approach.

Home range models provide ideal descriptive tools for conservation and management considerations based on basic spatial distributions of species; however, understanding the processes that govern movement and distribution allow greater biological inference such as predicting how animals may react to habitat loss or climate change (Beyer, Haydon et al. 2010). By combining observations of spatial distribution with descriptive physical and ecological habitat parameters at animal locations, selected habitat can be estimated based off a resource selection function contrasting used and available habitat (Boyce, Vernier et al. 2002). By identifying habitat types either “avoided” (unused) or “selected” (used) by an animal, it is possible to infer essential details regarding an organisms perception (and use) of its surrounding environment (Beyer, Haydon et al. 2010).

When resource usage within a home range is examined, a particular habitat unit is considered to be “selected” if it is used disproportionately to its availability (Alldredge and Griswold 2006; Beyer, Haydon et al. 2010). Habitat selection is considered to be distinct from habitat preference; which arguably can only be accurately determined by means of enclosure experiments, (and/or selection trials) which provide an equal availability of all habitats, but an important assumption of habitat selection studies is that preference can be

interpreted from observed patterns of use by natural populations (Garshelis 2000; Alldredge and Griswold 2006) Due to the heterogeneous distribution of resources in natural habitats, habitat preference (use of a habitat relative to its availability in the surrounding environment) is dependent on the availability of all surrounding habitat types (Aarts, MacKenzie et al. 2008 ). Therefore, inferences drawn from resource selection studies are influenced by the subjectivity in defining the limits of what is available to an animal (Beyer, Haydon et al. 2010).

Boyce (2006) defines resource selection functions as any function which is proportional to the probability of use of a resource unit. Habitat selection is a behavioral consequence of individuals actively selecting optimal habitat or passively persisting in certain habitats, and typically the extent of use within a particular area suggests the quality and abundance of resources in those areas (Boyce and McDonald 1999). RSFs are valuable tools to quantify resource use when animal distributions across landscapes may be linked to descriptive habitat coefficients at those used locations (Boyce, Vernier et al. 2002).

A common sampling design in RSFs estimated from presence-only radiotelemetry data involves contrasting a sample of habitat locations where a species is known to occur with a randomly generated sample of “available” locations drawn randomly within the domain of the study area (Boyce 2006; Johnson, Nielsen et al. 2006). If a habitat is used more than expected relative to its abundance across the landscape, it is assumed to be selected. If a habitat is used less than expected given its abundance, it is assumed to be avoided (Koper and Manseau 2012). RSFs are generally assumed to have an exponential or log-linear structure:

$$\text{RSF} = w(x) = \exp(\beta_1 x_1 + \beta_2 x_2 + \dots + \beta_k x_k)$$

for a vector ( $x$ ) of  $k$  predictor covariates; whereby the model coefficients ( $\beta_i$ ) for each corresponding habitat coefficient ( $x_i$ ) can be estimated from logistic regression (Boyce, Vernier et al. 2002; Manly, McDonald et al. 2002).

### **Modeling longitudinal data with Generalized Estimating Equations (GEE)**

Spatial habitat datasets are often burdened with spatial and temporal autocorrelation resulting from sequential positional records on individuals over time. This problem is particularly evident when employing GPS-based tracking capable of both high spatial and temporal resolution (*i.e.* more frequent observations of greater spatial precision than traditional radio triangulation (Boyce, Vernier et al. 2002; Dormann, McPherson et al. 2007)). In these telemetry surveys, spatial records can be correlated between time intervals of up to 1-month (Koper and Manseau 2009), and further autocorrelation exists due to small numbers of collared individuals, and intensive tracking of those few over time (Koper and Manseau 2012). Violating the assumption of the independence of observations due to autocorrelation increases the likelihood of type 1 errors - falsely rejecting the null hypothesis of no effect (Dormann, McPherson et al. 2007) Three typical responses for addressing autocorrelation include: (1) data censoring or rarefaction through sub-sampling in an attempt to achieve independence, (2) variance inflation to inflate standard errors following parameter estimation; or, (3) explicitly modeling alternative correlation structures within the statistical framework of analysis (Fieberg, Matthiopoulos et al. 2010) In the case of correlated data arising from longitudinal animal studies, Generalized Estimating Equations (GEE) are a convenient and general approach to modeling population-level inferences towards habitat use

under a binary ( or binomial - *i.e.* used (1) vs. available (0) habitat) distribution of the response variable (Koper and Manseau 2012). Empirical (also known as robust) standard errors are specifically included in model fitting to account for high levels of correlation between sequential positional records.

GEEs are a common approach when modeling highly correlated data from longitudinal studies in ecology and medical sciences (Hanley, Negassa et al. 2003; Ghisletta and Spini 2004; Cui 2007; Koper and Manseau 2009) as they allow the user to define correlation structure of sequentially collected datasets. In brief, the GEE (as an extension of the GLM) specifies how the population average of a response variable changes with covariates, while allowing for the serial correlation of repeated measurements of individuals over time. This focuses the estimation of regression parameters towards a population level interpretation, while treating correlation structure as a nuisance variable (Weiss 2005; Cui 2007). Other approaches; such as Bayesian spatial modeling, allow the flexibility to incorporate corrections for other complications such as: observer biases, missing data, or alternate error distributions; however, their complexity and intensive processing requirements are major detractors when faced with the robustness and simplicity provided by the GEE framework (Dormann, McPherson et al. 2007).

For the application into GPS telemetry designs, Koper and Manseau (2012) recommend GEEs incorporating both random effects as well as utilizing empirical (robust) standard errors. Random effects are included to account for the fact that data points arise from different animals, and those points within animals are not independent of one another (thus individual animal ID becomes a random variable). Empirical standard errors are necessary to account for autocorrelation due to the fact that specific correlation structures

cannot be modeled universally for the entire “used/available” dataset. This is due to the fact that the “available” portion is randomly generated and does not follow the same correlation structure as sequentially collected spatial fixes. In other words, it is advantageous to correctly model the correlation structure in order to increase the power of the analysis, but impossible when telemetry data is compared to random points. Thus the necessity of empirical standard errors (Koper and Manseau 2009; Koper and Manseau 2012).

RSFs fitted using GEEs and robust standard errors are considered marginal (population-based). Therefore, the interpretation of parameter estimates is population-specific, and model fitting using these methods is only appropriate within investigations that sample a subset of a population to determine overall responses rather than when attempting to determine a particular individual’s behavior over time (Koper and Manseau 2009; Koper and Manseau 2012).

#### *Spatial and Temporal Considerations*

Scale of selection is an important consideration when designing resource selection studies. The relative importance for specific criteria of selection will often differ when examined under the geographic range of a species, individual home range, or patch level selection (Allredge and Griswold 2006; Boyce 2006; Beyer, Haydon et al. 2010). Selecting an appropriate and biologically relevant scale of measurement is also essential. Choosing the incorrect scale of measurement may confound the degree of specialization or generalization of the study animal or population (Allredge and Griswold 2006) by obliterating spatial heterogeneity if resolution is too large, or under sampling habitat variance if the sampling of spatial extent is too small (Boyce 2006). Limitations of spatial data obtained from remote

methods (*i.e.* satellite or aerial photos) is often the determining factor in defining the extent and precision of spatial scales in resource selection studies (Minton, Higuchi et al. 2003). The minimum unit of selection cannot be smaller than the weakest resolution in a remote sensing or GIS database.

Ignoring the biology and life history of study species can also confound the interpretation of resource selection studies as it may overlook the effects of: season, gender, age, behavior, territoriality and diurnal cycles (Alldredge and Griswold 2006). Furthermore; in systems which display a disparity in resource use and/or availability across seasons, selection studies should be limited to shorter durations as pooling data across time may result in erroneous interpretations (Schooley 1994; Arthur, Manly et al. 1996; Cooper and Millsaugh 1999; Alldredge and Griswold 2006).

### 3 Habitat Selection

#### 3.1 Introduction

Anthropogenic development is a mounting pressure reducing natural habitats on a global scale, while increasing the number of wild populations living within or alongside urban areas (Rubin, Boyce et al. 2002). For many species, urbanization is equivalent to habitat loss and these animals respond to human activity through short-term flight (Andersen, Linnell et al. 1996), partial or temporal shifts in activities (Colescott and Gillingham 1998), or complete avoidance and abandonments of home ranges (Richardson and Miller 1997). Each level of this spectrum of avoidance can have important effects on foraging efficiency and fitness (Rubin, Boyce et al. 2002; Manor, Saltz et al. 2005). For example, several elk herds; when disturbed by humans during calving, have demonstrated a significant decline in reproductive success,(Phillips and Alldredge 2000) while mountain gazelle have been found to avoid altered habitats and exhibit greater flight distance with increased human presence (Manor, Saltz et al. 2005). At a molecular level, habitat fragmentation may reduce genetic diversity in bighorn sheep due to losses in connectivity and gene flow among populations (Epps, Palsbell et al. 2005).

In some systems, populations tolerant to frequent disturbances can adapt their foraging behavior and continue to use these disrupted habitats (Neuhaus and Mainini 1998) and experience greater fitness (Rubin, Boyce et al. 2002). Large mammalian predators are typically highly sensitive to anthropogenic disturbances due to their large home ranges and foraging strategies, and tend to avoid areas disturbed by human recreation and development; thus creating refugia for prey species that are able to exploit these altered habitats (Van Dyke, Brocke et al. 1986; George and Crooks 2006). North



American elk in the Bow Valley have exploited these urban refugia on a non-migratory basis for several decades and this pattern has begun to perpetuate in several wild ranging elk herds of Ya Ha Tinda Ranch (Hebblewhite, Merrill et al. 2006) and Rocky Mountain National Park (Lubow, Singer et al. 2002) , among others (Thompson and Henderson 1998).

While these behavioral strategies often maximize immediate fitness as urban environments present advantages both as refugia from predation and as sources of high-quality forage (*i.e.* cultivated or landscaped vegetation), the long term viability of these tactics remains unclear (Thompson and Henderson 1998; Rubin, Boyce et al. 2002). Many studies on urban wildlife concentrate on the issues of human-wildlife conflict (Messmer 2000), or management implications (Thompson and Henderson 1998; Adams 2005). There is still a lack of detailed studies examining resource selection within urban habitats and how this behavior affects wildlife and biological processes in fragmented landscapes (Rubin, Boyce et al. 2002). Understanding population distributions and behavioral responses (*i.e.* scale-dependent resource selection) to human disturbance and habitat fragmentation are important aspects to designing ecological research and monitoring programs and are essential when shaping wildlife management and land-use strategies (Boyce, Mao et al. 2003).

Resource Selection Functions are effective tools for examining complex spatial relationships between organisms and their available habitat. They have proven useful across comparatively simple examinations of single species distributions, such as in the case of Johnson *et al.* (2004) mapping patch and landscape distribution of mountain caribou (*Rangifer tarandus*), as well as complex relationships between sympatric species

(elk/mule deer spatial overlap - (Johnson, Kern et al. 2000), or predator-prey systems (elk predation following the re-introduction of wolves (Mao, Boyce et al. 2005). With the exception of several international studies utilizing RSFs; for example, behavioral adjustments in African herbivore responses to predation risk (Valeix, Loveridge et al. 2009), or wolf (*Canis lupus*) range expansion in Italy (Massolo and Meriggi 1998), the bulk of RSF development has been through modeling of North American ungulate and predator systems within the Rocky Mountains (Hebblewhite, Merrill et al. 2005; Frair, Merrill et al. 2007; Chetkiewicz and Boyce 2009), Yellowstone (Boyce, Mao et al. 2003; Fortin, Beyer et al. 2005; Mao, Boyce et al. 2005), and Great Plains (Sawyer, Nielson et al. 2007; Brook and McLachlan 2009) ecosystems.

The focus on North American predators within the body of RSF literature has largely concentrated on wolf populations and their potential effects on prey (predominantly elk) distribution (Anderson, Turner et al. 2005; Hebblewhite, Merrill et al. 2005; Mao, Boyce et al. 2005; Hebblewhite and Merrill 2007). Other studies have utilized large carnivore species as a basis to test conceptual methods of integrating RSFs with additional spatial analysis such as RSF and home range integration using grizzly (*Ursus arctos*) ranging data (Christ, Hoef et al. 2008), or Chetkiewicz and Boyce's integration of RSF and Least Cost Path (LCP) analysis to identify conservation corridors for grizzly and cougar (*Puma concolor*) in the Rocky Mountains of Alberta (Chetkiewicz and Boyce 2009).

In research pertaining to anthropogenic effects, wolf movement data has been used to model habitat selection as a function of human activity (Hebblewhite and Merrill 2008). Along similar goals, several North American ungulate populations have also given

ample opportunity to test and develop means of quantifying human mediated effects. RSFs have been used to investigate habitat use by bighorn sheep in urban environments (Rubin, Boyce et al. 2002), mule deer responses to resource distribution following oil & gas development (Sawyer, Nielson et al. 2006), and to model the risk of transmission of infectious disease between elk and cattle (Brook and McLachlan 2009).

Throughout the current body of literature, a large proportion of RSF development occurred using spatial data obtained through radiotelemetry on elk. Several authors dominating this body of literature include Dr. M. Boyce, Dr. E. Merrill, Dr. M. Hebblewhite, and Dr. C.J. Johnson whose numerous collaborations have utilized elk herds in Alberta (Ya Ha Tinda and Banff) and the United States (Yellowstone) to investigate the effects of scale in RSF habitat modeling (Boyce, Mao et al. 2003), mortality risk in translocation initiatives (Frair, Merrill et al. 2007), predator-prey dynamics (Hebblewhite and Merrill 2007), in addition to several conceptual papers aiming to improve RSF model predictions through refined approaches to model design and alternate error structures (Boyce 2006; Johnson, Nielsen et al. 2006; Hebblewhite and Merrill 2008). These authors have examined habitat selection by local Rockies elk populations on several occasions (Hebblewhite, Merrill et al. 2006; Chetkiewicz and Boyce 2009), establishing a strong theoretical basis for populating models with relevant physical and biological parameters. While this existing material provides the background for candidate model population, habitat selection has been repeatedly proven to differ greatly with scale (Boyce, Mao et al. 2003; Rumble and Gamo 2011); thus, final model interpretation should allow greater precision and site-specific reliability than the existing investigations due to the focused spatial scope of this study. Greater site specific

reliability will make resulting models better tools for predicting animal behavior and determining best land-use and management policies.

The primary goal of this investigation was to utilize high-resolution GPS collar data to model elk distribution in local urban and rural (wild) habitats at two spatial scales: a broader home range level analysis, and a more precise modeling of patch level selection within respective home ranges using Resource Selection Functions. I hypothesized that the extent of home ranges, while largely shaped by the local topography, will also be influenced by local anthropogenic effects. I predicted that both total home range area and “core” (*i.e.* 50% isopleth) would be smaller in urban herds due to fragmented habitats with inaccessible developed terrain (industrial & residential) and a small area of attractive artificially created foraging patches. Conversely, I predicted that the core and total home range in rural elk would be larger due to the distribution of animals over a broader natural landscape with fewer sources of anthropogenic fragmentation.

Within the context of Resource Selection Functions, I hypothesized that the physical and biological parameters influencing habitat selection should differ between rural and urban elk. I predicted that urban elk should select for artificial grazing sites and against rugged topography (steep slopes & higher elevations) and typically important cover habitat (due to the refugia effect). Rural elk were predicted to select largely in favor of habitat offering both cover and foraging potential (*i.e.* shrub, aspen, & spruce forest) while avoiding anthropogenic disturbances and rugged terrain.

## 3.2 Methods

### Study Site

The Bow Valley corridor is located 100km West of Calgary, Alberta, in the front ranges of the Canadian Rocky Mountains (Figure 3-1) and is characterized by rugged mountainous topography (1320m to 2972m). The climate consists of lengthy, cold, and dry winters interrupted by periods of warm weather due to Chinook winds; and relatively short, dry summers with the majority of summer rainfall occurring between May and July. This study focused on three non-migratory elk herds which reside in spatially distinct ranges within habitats along low elevation valley bottoms (2-5km in width), containing the highest quality habitat for elk (Paquet, Wierzchowski et al. 1996; Hebblewhite 2000).

Vegetation assemblages in the low-montane region consist mainly of lodgepole pine (*Pinus contorta*) forests with a distribution of Engelmann spruce (*Picea engelmannii*), white spruce (*Picea glauca*), willow (*Salix* sp.) and aspen (*Populus tremuloides*), combined with abundant parkland and grassland communities. Upslope subalpine regions are comprised mainly of Engelmann spruce, willow-shrub riparian zones, and grasslands which progress vertically to open shrub-forb meadows in high-elevation alpine communities (Paquet, Wierzchowski et al. 1996). Six species of ungulate are present in the Bow Valley ecosystem: elk, white-tailed deer (*Odocoileus virginianus*), mule deer (*Odocoileus hemionus*), moose (*Alces alces*), bighorn sheep, and mountain goat (*Oreamnos americanus*), and these support several important large carnivore species including cougar (*Puma concolor*) and wolves (*Canis lupus*) (Hebblewhite 2000).

This region of the Rocky Mountains is a major East-West transportation route and tourism hub. Valley bottoms are fragmented by three major highways (Figure 3-1 – Highways 1, 1A & 40), the national railway, preventative wildlife fencing along roadways and within municipalities, two major town sites (Banff & Canmore), several smaller hamlets, and a multitude of secondary roads and recreational developments (trail systems, ski resorts, golf courses, off-leash dog parks) (Hebblewhite 2000). From east to west the range of my study extended from Bow Valley Provincial Park, through the hamlet of Deadman’s Flats, and encompassed the town site of Canmore including a portion of Banff National Park along the southeastern park boundary.

Previous tracking and aerial survey of resident animals by Alberta Parks staff has established estimates of home range boundaries of three spatially distinct herds (Bow Valley Provincial Park – BVPP; Deadman’s Flats; and Canmore town site) and animal captures were targeted within these spatial boundaries with emphasis on equal sampling between town elk and wild herds in BVPP and Deadman’s Flats. Recent estimates suggest an average herd size of 80 elk for each of these locations (ATPR – John Paszcowski, Personal Communication).

### **Capture and Telemetry Protocols**

Between October 2008 and March 2009, 18 mature female elk were heli-netgunned, free-range darted or corral trapped (Table 3-1), and fitted with store-on-board global positioning system (GPS) radio collars (Telonics Inc., Mesa, Arizona, USA) equipped with VHF radiotransmitters and remote release mechanisms programmed to release after a period of one year (release date April 01, 2010 –Table 3-2). Collars were

programmed with an hourly fix cycle to provide the highest possible resolution of spatial movements while considering battery longevity for the study duration (Frair, Fieberg et al. 2010). An additional three collars were deployed via heli-capture January 2011 and recovered December 2011. All animal capture and handling protocols adhered to the University of Calgary Animal Use Protocol (Protocol #BI 2007-25).

During capture, the collars were distributed as widely as possible across the estimated ranges of the targeted groups in order to sample animals less likely to congregate into social groups and result in spatial correlation between individuals. All collared individuals were fitted with numbered eartags to allow for distance identification as well as identification after the collars released. Only mature females were radiotagged in order to maintain consistency of observations within the context of female habitat selection, as sexual differences in behavior and habitat selection are prevalent in gregarious sexually dimorphic ungulate species (Ruckstuhl and Neuhaus 2002). The potential for male dispersal also raised concerns of irretrievable collar loss in the case of permanent long range migration and consequently males were omitted from this study.

Collar longevity ranged from 1-month to slightly over one full year, life span being limited by several mortalities and accidental collar loss (Table 3-2). Collared animals were located on site approximately once every two weeks in order to collect behavioral and demographic data, for fecal sampling, and to allow periodic verification of collar status and operation (Beringer, Hansen et al. 1996). Despite the advances in GPS telemetry allowing the accurate collection of fine scale spatial data, behavioral data is necessary to complement this in order to quantify survival and reproductive success, link behavior to movement, and understand habitat use in the context of fitness and survival

(Beyer, Haydon et al. 2010). All collars were equipped with mortality sensors that activated after 24 hours of immobility. The probable cause for each mortality (*i.e.* vehicle strike, predation, or unknown) was determined by the condition of animal remains and presence of predator signs. There was no indication of any animals succumbing to capture myopathy in the weeks following collar deployment (Beringer, Hansen et al. 1996).

Data retrieved from GPS collars was first cleaned by removing unsuccessful fixes, extraneous readings inadvertently obtained before or after deployment cycles, as well as restricting final datasets to 3-dimensional fixes in order to preserve data integrity and reduce bias due to positional error (Frair, Fieberg et al. 2010). Fix-rate bias was not an issue due to the high final fix rate (overall 92% of all hourly fixes were retained), and I did not correct GPS locations as all used data was high precision three dimensional data (Table 3-1). Data from several collars was discarded due to unacceptable data loss occurring from collar malfunction (Table 3-2). Cleaned data was imported into a geographic information system (GIS) database (ArcGIS 9.3 & 10.1, ESRI 2012) and projected using a Universal Transverse Mercator (UTM) coordinate system for further processing utilizing several commercial and open-source programs (See Appendix A).

### **Independence of movements among radio-collared elk**

Before any detailed analysis occurred, spatial cohesion between individuals within each herd was examined by time-matching all possible animal pairs ( $n_{\text{Canmore}}=21$ ,  $n_{\text{BVPP}}=15$ ,  $n_{\text{Deadman's}}=10$ ) and determining distances between individuals whenever GPS fixes coincided as per Fortin *et al.* (2005). Animals greater or equal than 200 meters apart



were considered to be independent while those closer than 200 meters were considered to be part of the same social/foraging unit, based on patterns of herd behavior and dispersal during field studies (Table 3-3, Table 3-4, Table 3-5).

### **Home Range: $\alpha$ -LoCoH**

In order to capture the potential effects of linear developments and anthropogenic disruptions within this study area, I chose to utilize the methodology developed by Getz *et. al* (2007), and the LoCoH toolbox produced by the same authors for ArcGIS 9.3 (See Appendix B – Software & Open-Source ArcGIS Tools), and estimated home range for collared individuals using the  $\alpha$ -LoCoH procedure. Estimates for the determining variable,  $a_1$ , were initially selected according to the “rule of thumb” suggested by Getz *et al* (2007) whereby  $a_1$  is equal to the maximum distance between any two points in the data set. Further model selection progressed via the “minimum spurious hole covering” (MSHC) rule, which was facilitated due to known habitat voids in local topology where elk presence was not possible due to fenced compounds, linear barriers, large anthropogenic structures, and hydrological/geological barriers such as lakes and mountain peaks. Using the “rule of thumb” as an initial draft, the defining factor ( $a_1$ ) was increased (or decreased) in such a manner that spurious holes in the home range were covered but expected habitat voids remain excluded from the final home range estimate. This method is arguably subjective to the interpretation of the analyst, however it has been demonstrated that the value of  $a_1$  is highly robust to misspecification. In trials utilizing simulation data, these heuristic rules provided estimates which were (on average) greater or lesser than the actual value of  $a$  by a factor of only 0.2 (Getz, Fortmann-Roe *et al.* 2007).

Radiotelemetry and observations over the duration of the field season suggested that the elk herds were variably cohesive dependent on season, thus home ranges were constructed for each individual based on a full telemetry dataset and compared to pooled data home ranges constructed for each of the BVPP, Deadman's Flats, and Canmore herds. At first, home ranges were constructed separately by season (winter & summer) but displayed no significant differences, so final models included pooled data from both winter and summer. Pooled data was randomly rarefied to consist of only 50% of the original telemetry points. Rarefaction was necessary to reduce dataset size to accommodate computer processing capability, as analysis upon the full dataset exceeded the capacity of our systems. Home range limits were retained at the 95% isopleth level due to the high quality of telemetry data, small home range sizes and the observed absence of seasonal migratory behavior in all collared individuals.

Finally, MCPs were calculated using Hawth's Spatial Analysis Tools in order to provide a comparison against  $a$ -LoCoH estimates, as well as for use in RSF protocols (Figure 3-2).

### **Resource Selection Function**

Resource selection functions were estimated using the used-available sampling design as per (Johnson, Nielsen et al. 2006) and (Boyce and McDonald 1999) while using GEEs to generate conditional (herd-specific) inferential models as per Koper & Manseau (2012). Descriptive habitat parameters were selected based on biological relevance (as established in previous RSF studies on elk and similar ungulates), relevance to local

natural and anthropogenic pressures, and availability of pertinent remote sensing data. (Table 3-6 & Table3-7)

All telemetry locations were pooled by herd. Animal locations were not directly controlled for inconsistent sample sizes across animals; thus, model inferences to each herd may be biased to elk with longer telemetry records. However, I used GEEs with empirical standard errors to account for autocorrelation within sampling units (*i.e.* individual animals) and included grouping of observations within individuals as random effects. GEE's used in conjunction with empirical (robust) standard errors produce estimators which are robust to deviations in sample sizes (Koper and Manseau 2009).

#### *Used versus available sites & correlation structure*

Accurately identifying “used” habitat is a major difficulty in spatial or resource selection studies, the use of modern radiotelemetry and GPS devices allows the collection of continuous high-resolution spatial data which eliminates a large degree of uncertainty in defining these used habitat units when fix success is high and/or habitat-related data loss (*i.e.* dense canopy cover) is low (Nielson, Manly et al. 2009; Frair, Fieberg et al. 2010). Samples of “available” habitat can be subject to contamination (*i.e.* “available” habitat sites are actually sites observed to be “used”) as sampling intensity increases in GPS or telemetry monitoring field programs. Johnson *et al.* (2006) determined that; while overlapping distributions of “used” and “available” habitat units does typically occur, sampling designs are robust to contamination.

Using a home-range based approach (Johnson and Gillingham 2008), I calculated a 100% minimum convex polygon (MCP) home range for each herd, and (without

replacement) randomly drew the same number of locations as were in the “used” telemetry dataset within the boundaries of the MCP to make up the “available” dataset (Figure 3-3). This scale of selection limits the scope of interpretation to habitat normally available within a static home range (Koper 2009). The generation of points for the “available” dataset results in an inability to correctly model correlation structures uniformly across the entire dataset as the “used” points are likely to display spatial and/or temporal correlation, while the “available” portion of the dataset are random. Fortin *et al.* (2005) developed RSF-type models with estimating equations using an independence-working correlation structure and empirical (robust) standard errors, which is suitable when assessing habitat selection at a local scale (Koper and Manseau 2009).

Decreasing the time interval between GPS location inevitably increases the level of autocorrelation in the resulting data, by which each individual location is increasingly influenced by the locations preceding it in time (Frair, Fieberg *et al.* 2010). Failure to account for this correlation within subjects (*i.e.* radio collared individuals) can result in erroneous estimates of regression parameters, especially so when correlation is strong (Ballinger 2004). Generalized estimating equations (GEE) are often utilized in longitudinal or repeated measure designs to provide unbiased parameter estimates from highly correlated or non-normal response data without the need for data transformations (Ballinger 2004). GEE parameter estimates with empirical standard errors are robust to misspecification of the correlation structure and generate consistent parameter estimates when the mean response is correctly specified (Cui 2007); this is necessary when comparing correlated (“used” telemetry data) with randomly generated (“available”) datasets (Koper and Manseau 2012).

Applying robust standard errors with working correlation structures other than independence have not been previously used in RSF studies, however they may help overcome biases towards data-rich animals and so I attempted to estimate RSF parameters using both independent and compound symmetric working correlation structures (Koper and Manseau 2009; Koper and Manseau 2012). Many of the models utilizing compound symmetry as the working correlation structure failed to converge and so all final models reported in this study were constructed using an independence working correlation structure, but report parameter estimates using a robust (empirical) error structure rather than model-based standard errors.

#### *Habitat Parameters*

Many recent elk-specific RSF studies have made liberal use of openly accessible, high-precision remote sensing data to extract relevant physical/geographical and biological predictor variables for use in model construction. In general, descriptive characteristics used in model construction within the scope of current North-American literature fall into three categories: topographic; landcover/vegetation; quantitative food resources; and anthropogenic (Johnson, Kern et al. 2000; Hebblewhite, Merrill et al. 2005; Chetkiewicz and Boyce 2009; Coe, Johnson et al. 2011). Topographic and physical landscape variables in most RSF investigations include slope, elevation, aspect, or some combination of several as a terrain ruggedness index, and distances to permanent water (Chetkiewicz and Boyce 2009) or snow-water equivalents (Mao, Boyce et al. 2005). Anthropogenic variables typically include distance to roads or trails (Hebblewhite, Merrill et al. 2005; Sawyer, Nielson et al. 2007), distance to roads by user intensity (Johnson, Kern et al. 2000), average road density (Chetkiewicz and Boyce 2009), or

length of road segments within specific buffers of animal observations (Rumble and Gamo 2011). The scope of quantitative food resources include specific vegetative or forage biomass production or greenness (Coe, Johnson et al. 2011), and predator (Anderson, Turner et al. 2005) or prey (Chetkiewicz and Boyce 2009) encounter probabilities (usually based on a separate RSF prediction).

Landscape variables are usually extracted from remote sensing aerial or satellite photography or established vegetation inventories (*i.e.* AVI) and in recent publications have included a number of directly extracted landcover classes including, but not limited to: grasslands, shrubs, conifer stands (open and dense), deciduous stands (open and dense), alpine, bare rock, ice/snow, historic and recent burns, and avalanche paths; as well as landcover variables calculated from the AVI which include: circularity of patch, patch diversity, distance to edges, canopy closure, distance to cover and distance to forage (Johnson, Kern et al. 2000; Hebblewhite, Merrill et al. 2005; Mao, Boyce et al. 2005; Sawyer, Nielson et al. 2007; Chetkiewicz and Boyce 2009; Coe, Johnson et al. 2011; Rumble and Gamo 2011)

I selected a number of habitat variables known from previous studies to be important and relevant predictors, in order to build candidate model sets for the Canmore and BVPP/Deadman's Flats herds (Table 3-6 & Table 3-7) (Fortin, Beyer et al. 2005; Hebblewhite, Merrill et al. 2005; Sawyer, Nielson et al. 2006; Sawyer, Nielson et al. 2007; Rumble and Gamo 2011).

Landcover (vegetation and habitat type) parameters were calculated from the most recent Alberta Vegetation Inventory (AVI - (ASRD 2005)) which was manually updated

based on 2009 aerial photography supplied courtesy of Scott Jevons (ATPR), to reflect changes in anthropogenic disturbance within town sites (golf course expansion and residential development). Road proximities, elevations, slopes, and hydrography were calculated from federally regulated datasets such as Canadian Digital Elevation Data (CDED), National Hydro Network (NHN), and National Road Network (NRN), available through the Canada Council on Geomatics (CCOG) ([www.geobase.ca](http://www.geobase.ca)). Parameter values were calculated using tools provided in the Geospatial Modeling Environment (Beyer 2012) for arcGIS 10.1 (ESRI 2012).

Variables related to topographic and geographic factors included slope (*Slope*), elevation (*Elev*), and distance to permanent water (*Hydro*).

Anthropogenic factors included: distance to golf course (*GC*), length of secondary roads within a 100m buffer around each used or available location (*Road*), and area ( $m^2$ ) of anthropogenically modified non-forage land cover classes within a 100m buffer around each used or available location (*LC2*). *LC2* cover types were derived from the Alberta Vegetation Inventory and included: non-vegetated gravel pits/borrow pits (AVI code: AIG), industrial/plants (AII), non-vegetated surface mines (AIM), cities/towns/villages/hamlets (ASC – excluding potential grazing terrain within municipal limits), and rural development (ASR).

Other landcover variables included: area ( $m^2$ ) of trembling aspen within 100m buffer of points (*V1* - defined as aspen monocultures or mixed forest with >60% aspen), area ( $m^2$ ) of lodgepole pine within 100m buffer of points (*V2* - monoculture or >60% pine in mixed forests), area ( $m^2$ ) of white spruce within 100m buffer of points (*V3* -

monoculture or >60% pine in mixed forests), area (m<sup>2</sup>) of forage class land cover (including anthropogenic sources) within 100m of points (*LC1*), unsuitable habitat (*LC3*), and shrubs (*LC4*). Forage class landcover was defined as: farmsteads (*AIF*), non-vegetated permanent right of ways (*AIH* – paved surfaces within right of ways were not excluded due to negligible overall contribution to area calculations), vegetated right of ways (*CIP*), reclaimed/vegetated gravel pits and mines (*CIW*), and pastureland including anthropogenic grasslands/golf courses (*CP*), and herbaceous grassland (*HG*). Unsuitable landcover classes included: cutbanks (*NMC*), barren rock (*NMR*), flooded areas (*NWF*), and permanent ice/snow (*NMI*). Shrubs were defined as non-forest (<6% treed) open (*SO*) and closed (*SC*) shrubland.

Model construction from these basic parameters followed the AIC premise of biological relevance utilizing field observations and literature support to define a candidate set of 15 models (Table 3-8). Model construction was based on several approaches: (1) Utilization of previous “best” selected models (*i.e.* Anderson *et al.* (2005) and Chetkiewicz & Boyce (2009)) for base models which were modified to reflect additional regional specific parameters, (2) biologically relevant models based on prominent features and vegetation classes in target regions (*i.e.* BVPP Model and Canmore Model), or (3) biologically relevant models constructed to reflect various aspects of elk foraging and resting behavior.

In constructing biologically relevant models, approximately half of the candidate set concentrated on habitat selection based on purely biological or anthropogenic effects, while omitting the effects of topography which included elevation and slope. Elevation and slope are primary determinants of habitat use for elk (Toweill and Thomas 2002),



however the rationale was that restricting the pool of randomly generated “available” habitat locations to within the MCP of the telemetry dataset; and thus restricting it to valley bottom home ranges, would result in very minimal elevation and topographic differences among all “available” locations. These models (Table 3-8: Model Number 1 to 7) would therefore strongly predict habitat selection purely based on aspects of elk behavior related to grazing and browse selection, cover for thermoregulation and predator evasion, and attraction or avoidance of anthropogenic features. The remaining candidate models (Table 3-8: Model Number 8 to 15) took the opposite approach and predicted that differences in slope and elevation; even across a small landscape scale within the valley bottom, would influence habitat selection significantly in addition to the biological, landcover, and anthropogenic effects discussed above.

#### *Season*

Summer was defined as May-September, and winter was defined as November-April, as per elk RSF application by Anderson et al (2008). Datasets were segregated by season and models fit independently to each subset thus recognizing the potential for differential habitat use across seasons (Schooley 1994; Arthur, Manly et al. 1996; Cooper and Millspaugh 1999; Alldredge and Griswold 2006). Due to the influence of male herding behavior during the rut, telemetry data during the rut (late September-October) was omitted from this analysis to avoid confounding resource selection behavior between the sexes.

#### *Model Selection*

Using the habitat parameters listed above as predictor covariates, I estimated a RSF under an information theoretic (Anderson and Burnham 2002) approach, using GEEs and 15 *a priori* RSF models determined to be biologically meaningful (Table 3.8 & Table 3.9). Goodness of fit between competing models was assessed by their quasi-likelihood information criterion score (QIC – Pan 2001). Relative support for each model was considered by calculating differences in QIC ( $\Delta\text{QIC} = \text{QIC} - \text{minimum QIC}$ ) between each model and the lowest (best) score, such that models  $\pm 2.0$  QIC points of each other were considered to share similar support for goodness-of-fit and correctly approximating our observations (Anderson and Burnham 2002). To avoid multicollinearity, an estimated correlation matrix for all covariates was examined and only uncorrelated independent variables were analyzed together in individual models. (Anderson, Turner et al. 2005)

### 3.3 Results

Evaluation of concurrent locations among all possible pairs of radio-collared female elk suggests that elk throughout the Bow Valley were not behaving completely independently for the duration of the study and spatial correlation between animals is often but not always high (Table 3-3, Table 3-4, and Table 3-5). In Canmore, six of seven elk were highly cohesive with tight spatial distributions in 13-38% of summer and 47-83% of winter fixes, indicating the elk are moving within a single dynamic herd subject to fission-fusion cycles which modulate group composition. Only one elk (ID 125) was independent and, while it exploited terrain within the Canmore limits at the municipal golf course, it spent a large portion of time interacting with elk within Banff National Park and was not a static member of the Canmore herd.

All the collared animals in Bow Valley Provincial Park displayed consistently higher spatial cohesion during winter (53-75%) than summer (18-39%). Each animal within this region was considered to be a member of a single herd which displayed dynamic fission/fusion cycles over time; however, no animals were spatially independent of any others. Similarly, collared elk within the Deadman's Flats area had greater spatial cohesion in winter than summer, although high mortality and sparse sampling made the identification of social groups difficult.

### **Home Range**

Home range estimates for individual Canmore elk identified similar spatial extent and core areas (core home range was considered to consist of isopleths encompassing 50% of all telemetry fixes as defined by Rubin *et al.* (2002)) for animals considered to be year-round residents (1,2,4,32,123,124 – See Figure 3-4). These animals displayed use of high-quality foraging potential at the Canmore Municipal Golf Course, Silvertip Golf Course, Millennium Park, and the Three Sisters Golf Course Development (including dense coniferous forest down-slope of the golf course (See Figure 3-5)). Core home range areas (indicated by 20% and 50% isopleths) were concentrated around grassland-type habitat found on these golf courses, soccer and baseball pitches at Millennium Park, and throughout several green belts in peripheral residential neighborhoods. Core home range area was small, consisting of only 2.31 km<sup>2</sup> when the influence of elk 125 (non-resident) was excluded from a herd-level home range estimate (4.93 km<sup>2</sup> when data from all animals was pooled). The home range utilization distribution also indicates that the Canmore elk were utilizing riparian zones and mid-stream islands along the Bow River for cover and bedding habitat as well as movement corridors between important foraging

locations at Three Sisters to the South and the Municipal Golf Course to the North, this was consistent with observations during manual radio tracking.

Comparison of Canmore home range estimates against established wildlife corridors as determined by Herrero (2000) and Alberta Parks (2003), (Figure 3.6) showed an absence of use at several predicted habitat patches (*i.e.* Canmore Nordic Centre, Indian Flats, Grassi Lakes, & portions of the South Canmore habitat patch), while use of the established primary multi-species wildlife corridor was low with the exception of the down slope margin of the corridor where it bisects fairways at the Three Sisters Resort. Corridors allowing access between the Bow River and Three Sisters Golf Course were heavily used by elk (Grassi and Three Sisters Creek Secondary Corridors).

Individual home range estimates for collared animals in BVPP indicate similar home range extents for each animal, with some differences due to mortalities reducing the duration of fix cycles (Figure 3-7). A large proportion of the home range was within Bow Valley Provincial Park, bounded to the north and south by Highway 1A and the TransCanada while the eastern extent of each home range closely approximated the boundary of the Stoney 142-143-144 Reservation. The western portion of the home range extended towards the Hamlet of Exshaw where it was severely constricted from steep valley walls and terminated near the town site where physical barriers were greatest (Figure 3.8) The core home range area at a 50% isopleth was an estimated 4.2 km<sup>2</sup>, with a distribution of sites within BVPP and south of the TransCanada highway. Spatial voids within the home range included areas around the provincial park campground and parks maintenance and visitor services facilities, indicating BVPP elk are “avoiding” anthropogenic features and areas with high human-use.

Home range estimates for the Deadman's Flats herd had limited predictive ability due to the high early-stage mortality of collared animals. Only one animal was found to utilize the Wind Valley regional habitat patch (Figure 3-6 and Figure 3-9) but was killed by a cougar shortly after leaving the valley bottom. The remainder of collared animals were restricted to the North side of the highway at locations in the Bow Flats regional habitat patch and at Lac des Arcs (Figure 3-10). The animals within Deadman's Flats often selected a small patch of land isolated by Lac des Arcs and the Bow River, likely due to its remoteness and difficult access (Figure 3-11), although carcass counts in this area suggest that predation pressure is high from natural and human sources (a large number of hunting stands were present along with modified pathways and clearings intended to herd elk to facilitate bow hunting). Core home range area (50%) reflected the highly restricted nature of the valley at a size of 0.42 km<sup>2</sup>.

### **Resource Selection**

A total of 48 580 GPS locations (1909 – 11827 per elk – Table 3-1) were used to develop seasonal models for elk in Canmore based on 15 *a priori* candidate models derived based on biologically relevant construction (Table 3-9). The model with the greatest support based on QIC scores was identical for both Summer (QIC: 39799) and Winter (QIC: 60288) seasons, and no alternate models shared similar empirical support based on QIC (*i.e.* within 2 QIC score points of the top model), however, parameters such as slope, elevation, distance to water, local road length, and local forage habitat area are common in all top ranked models.

Based on best model estimates (Table 3-10), elk avoided steeper slopes and higher elevations. Distance to water had a slight positive effect on selection whereas road length within a 100m buffer had a negative effect on selection, indicating animals were avoiding higher density road networks and selecting areas further from water. Important effects of landcover included a negative effect of selection from white spruce stands, indicating animals were using white spruce in a proportion which was less than the availability across the landscape; while positive correlation of forb/grazing-type landcover with habitat use indicated selection of this terrain.

In BVPP/Deadman's Flats, a total of 36430 GPS Locations (254 – 10303 per elk – See Table 3-1) were used to develop seasonal models for elk outside of Canmore in BVPP and Deadman's Flats based on 15 *a priori* candidate models derived based on biologically relevant construction (Table 3-11). The model with the greatest support based on QIC scores was identical for both Summer and Winter seasons, and no alternate models shared similar empirical support based on QIC (*i.e.* within 2 QIC score points of the top model). Model estimates from the best supported summer and winter models indicated that elk were selecting against steep slopes and higher elevations and were selecting habitat in response to several important landcover classes (Table 3-12). White spruce and shrub landcover was selected for similarly in both winter and summer based on model estimates, whereas aspen landcover was selected for in both winter and summer with greater model estimates in summer. Model effects predicted the selection of forb/grazing landcover in both winter and summer seasons, with larger parameter estimates predicting selection during winter months.

### 3.4 Discussion

High spatial correlation between elk pairs suggests that the animals are sharing common areas throughout their respective home ranges and are associating within dynamic herd structures, resulting in similar spatial distribution and habitat use. The absence of cohesion between groups, as well as spatial segregation; especially at the Deadman's Flats/BVPP interface, suggests potential barriers to gene flow between these herds. The potential impact of genetic barriers warrants further investigation into the role of male dispersal in maintaining genetic diversity throughout the Box Valley.

#### Home Range

An important consideration when constructing unbiased home range utilization distributions is that data collection must occur often enough to obtain a representative sampling of points across all behaviors, if this is not achieved when the resulting utilization distribution may be biased against sparse behavior (Getz, Fortmann-Roe *et al.* 2007) I considered the high quality (>90% data retention) and high frequency (hourly) GPS data to have a good descriptive ability in detecting most behaviors of the collared animals, and unlikely to be significantly biased towards any specific activity. Moreover, the distribution of data points was dense relative to the mean hourly movements of collared individuals, allowing us to detect voids in the home range distribution as actual physical obstacles within the landscape.

Comparisons of home ranges based on area did identify differences in core home range areas as well as total (95% isopleths) home range size (Figure 3-12) which were largely consistent with our hypotheses. As predicted, core home range size was much

smaller in the Canmore herd (~2.3 km<sup>2</sup> at 50% isopleth) relative to the rural animals in Bow Valley (~4.2 km<sup>2</sup> at 50% isopleths) which is indicative of the selection of smaller, specific habitat fragments with more frequent use. Studies on voles (*Microtus oeconomus*), for example, reported responses to habitat fragmentation as increasing rates of congregation and higher animal densities with greatly overlapping home ranges on residual habitat patches (Ims, Rolstad et al. 1993). Furthermore, Rubin *et al.* (2002) reported that populations of female bighorn sheep utilizing urban areas in southern California had smaller core activity areas and selected lower elevations and gentler slopes. This is consistent with patterns of use in the highly fragmented Canmore region – where core home range was small and consisted largely of modified grazing patches while steep slopes outside of the valley bottom were avoided.

When investigating residency time by elk across patchily distributed resources, Anderson et al (2008) determined that wild elk in Wisconsin preferentially and repeatedly visited locations with high resource-selection values but their transit times across this terrain was unrelated to habitat quality, indicating the animals did not slow down in preferred areas. This contrasts with observations of Canmore elk herd which often utilized preferred high quality habitat as both foraging and bedding locations, especially in winter months when group sizes were large. Random residency times are likely a behavioral strategy to reduce predictability and lower predation risk (Anderson, Forester et al. 2008), and elk within the town site have adopted a tendency to linger on high-quality patches; such as the golf courses, identified within the “core” home range areas. The frequent congregation of animals within these reduced habitat patches has implications for density-dependent interactions including disease, pathogen, and parasite



transmission (Gillespie and Chapman 2006; Gillespie and Chapman 2008; Puttner, Meyer-Lucht et al. 2008). The effects of landscape on host-parasite assemblages are examined in Chapter 4.

Total home range size differed similarly, with larger total area in Bow Valley (95% isopleth: 22.4 km<sup>2</sup>) versus Canmore (95% isopleth: 13.7 km<sup>2</sup>), further supporting the hypothesis that urbanized animals in Canmore are persisting within reduced ranges focused on smaller residual habitat patches and maintaining close proximity to sources of anthropogenic disturbance. Total home range in the Deadman's Flats herd was similar to Canmore (95% isopleth: 10.8 km<sup>2</sup>), although this appears to be largely due to topography influencing distribution, and highway barriers limiting range expansion outside of the valley bottom. Animal sampling within the Deadman's Flats herd was sparse compared to the other locations, and it was suspected to have poor sampling of local animals utilizing terrain throughout the Wind Valley (south and upslope of Deadmans Flats).

Abiotic factors (*i.e.* topography, hydrology) are suggested to be the primary determinants of large-scale distribution of herbivores (Anderson, Turner et al. 2005). The impact of rugged mountainous topography was evident in all three herds throughout the eastern portion of the Bow Valley. Animal movements were restricted to valley bottoms and there was a prominent pinch point between the Deadman's and BVPP home ranges where steep slopes impinged a great distance into valley center and compounded barrier effects of the highways, wildlife fencing, and industrial development at this location (Figure 3-11). As evidenced by home range limits and an apparent absence of interaction between the Bow Valley Provincial Park and Deadman's Flats animals, this pinch point is currently acting as a significant barrier to longitudinal movement of elk along the Bow

Valley, and likely limits the East-West connectivity between Kananaskis and Banff National Park to movement corridors passing from Nakiska (along Hwy. 40) through the Wind Valley (which is currently subject to massive 10,000 unit development from the Three Sisters receivership, Price Waterhouse Cooper). Spatial constraints within this area make mitigation at this pinch point unlikely unless significant effort in constructing highway crossing structures is possible to mitigate the effects of fragmentation resulting from the TransCanada highway (Clevenger and Waltho 2005).

The barrier effect of the TransCanada highway is especially visible in the South Canmore region, which is a critical wildlife linkage for animals moving along the valley between the Wind Valley habitat patch and Banff National Park, and beyond (Herrero 2000). The imposition of the Three Sisters and Stewart Creek developments in this area; compounded with a non-functional wildlife corridor situated too far upslope in poor habitat areas (non-functional due to limited cover potential and steep slopes), is a significant disruption to wildlife movement and forage selection in the South Canmore region. Propositions to construct wildlife exclusion fencing around the perimeter of the Three Sisters development could have catastrophic effects on the local elk by eliminating a significant portion of the core home range area and restricting lateral movement between the Bow River and heavily grazed terrain at Three Sisters.

Herrero (2000) identified significant deficiencies of currently accepted movement corridors within the Canmore town site, identifying them to be largely “non-functional” due to limitations presented by insufficient widths to compensate for high slopes and hiding cover deficiencies, and from design being based on guidelines intended to be applied in non-winter months in remote forestry settings. The lack of functional

movement corridors within the Canmore town site may impair the movement of wildlife through the Bow Valley between Kananaskis Country and Banff, and is likely to have restrictive effects on the persistence of human-wary carnivore species (Herrero 2000). Very low rates of natural mortality in the Canmore animals, avoidance of cover-type habitats, and extended use of anthropogenic foraging habitats suggests that large carnivores are relatively absent from the municipal areas.

The current distribution of predicted habitat in the Canmore region is built around largely invalidated movement corridors, and several of these habitat patches are underutilized or avoided by the Canmore animals. While certain utilized patches do not adhere to technical specifications of acceptable patches (*i.e.* Bow Corridor Ecosystem Advisory Group (BCEAG) guidelines suggest: 1.2km wide, 4.5km<sup>2</sup> total area), distribution of resources and human activity has made some smaller areas attractive to the Canmore animals. Such an example is the cleared terrain at the Three Sisters development where a reduction of recent human activity and abundant forage has created a massive draw, whereas the Canmore Nordic Centre habitat patch (which falls at the Northwestern limit of the principal Canmore wildlife corridor) is avoided by elk, likely due to consistent annual recreational use.

Predation avoidance at a landscape scale through migration has been shown to reduce exposure to wolf predation by up to 70% in elk within Banff National Park, Alberta, while maintaining higher overall forage quality than resident animals foregoing migration by staying closer to human activity (Hebblewhite and Merrill 2009). Although urban-dwelling animals in that study had lower average forage quality, by using areas close to human activity they were able to forage more selectively in the absence of

predation risk, essentially reversing the typical positive correlation between high forage quality and predation risk (Hebblewhite and Merrill 2009). This highly successful alternate strategy was observed in the resident Canmore animals where predation mortality was nil in collared animals. The resident animals in Deadman's Flats and BVPP have also adopted a non-migratory behavior and established static annual home ranges, but are subject to less human activity and have correspondingly higher natural predation rates (3 of 11 collared animals were lost to predation).

### **Resource Selection Function**

Generally, home range utilization distributions for elk within the Canmore town site indicated they preferentially selected anthropogenically modified landscapes on the periphery of the main residential and commercial districts while avoiding the municipal core, rugged topography (*i.e.* high-angle slopes and terrain above 1500m) as well as several high-use recreational zones (*i.e.* Canmore Nordic Centre and an off-leash dog park). As expected, these home range estimates are consistent with RSF models which identified preferential use of forb/forage landcover within the town site as well as avoidance of high density road networks and avoidance of cover habitat types (*i.e.* white spruce) as well as high altitudes and steep slopes.

Elk typically select areas of high forage biomass to maximize net energy intake; however, they will also increase their use of forested areas in the presence of predators where the potential for cover lessens the cost of vigilance behavior (Anderson, Turner et al. 2005). Rubin *et al.* (2002) reported that bighorn sheep grazing near metropolitan areas had greater diet quality; similarly, Hebblewhite and Merrill (2009) reporting that greater

selectiveness by elk near urban habitats allowed higher forage quality than rural animals. Chetkiewicz and Boyce (2009) reported low habitat selection values and low probability of occurrence for large carnivores (cougar and grizzly bears) within the Canmore town site, and over a large portion of the core and annual home range selected by the town elk, supporting the interpretation that these animals are preferentially selecting high-quality forage over cover terrain, as the relative risk of predation is reduced when predator resource selection values are low.

Elk respond to predation pressure by shifting preferences in habitat; case studies in Wisconsin have found that under high predation pressure from wolves, elk preferentially chose conifer stands and this preference shifted to open grassland and as pressure decreased (Fortin, Beyer et al. 2005). Mao *et al.* (2005) reported that elk select habitats that help to avoid predation pressure in summer while relying more on other tactics (e.g., forming larger groups) during winter months. This apparent balance between forage and cover-type habitat was apparent in RSF values estimated for the rural elk ranging throughout BVPP and Deadman's flats and habitat selection was greatly influenced by selection for habitat classes offering cover as well as forage potential (*i.e.* shrub, aspen, and white spruce). Parameter estimates for winter and summer were similar for the static topographic effects of slope, whereas landcover class variables indicated greater selection for aspen and shrubs in winter (white spruce and forbs are still selected in greater proportion to their availability), while winter estimates indicate larger magnitude of selection for forb/grazing and shrubs (aspen and white spruce are still selected, but to a lesser magnitude). Model estimates indicated that animals were selecting both cover (shrub, aspen, and white spruce) and exposed foraging (forb/grazing)

landcover classes in greater proportion of their availability across the landscape. The overlap of increased selection for two spatially exclusive landcover types is best explained by daily temporal cycles in resource use through typical foraging patterns near dusk and dawn with bedding in cover habitat through the day. Further separation of datasets into temporally explicit model sets may increase the resolution of habitat selection estimates, and particularly when analyzing activities (obtained from activity switches on the collars) in specific habitat types.

I did not expect that model selection would identify the same model for both winter and non-winter months. Multiple studies have identified season as a critical element in determining both scale and direction of habitat selection in: large carnivores such as grizzly and cougar (Nielsen, Boyce et al. 2003; Chetkiewicz and Boyce 2009), wolf (Hebblewhite and Merrill 2008), mule deer (Sawyer, Nielson et al. 2006), moose (Dettki, LÅfstrand et al. 2003), bighorn sheep (Rubin, Boyce et al. 2002), caribou (*Rangifer tarandus*) (Johnson, Seip et al. 2004), and multiple studies involving elk (Johnson, Kern et al. 2000; Boyce, Mao et al. 2003; Mao, Boyce et al. 2005; Coe, Johnson et al. 2011). The most significant difference between our study and those mentioned above are that previous RSF studies; and especially those involving elk, were designed and validated over large geographic areas due to migratory or wide ranging behavior which resulted in home ranges of up to several orders of magnitude larger than the estimates for the Bow Valley animals. Spatial constraints due to the small non-migratory ranges of the Bow Valley animals likely reduces the potential for spatially explicit summer and winter ranges, or renders the resolution of current models unable to discern very subtle differences. The lack of migratory behavior in the Bow Valley

animals is one of many examples of declining migratory behavior on a global scale with similar examples seen in Alberta at the Ya Ha Tinda herd and elsewhere in the American Mountain Parks (Hebblewhite, Merrill et al. 2006). When migratory action ceases, winter range can begin to approximate summer range, and seasonal variations in behavior might vanish.

Overall, the elk within the Bow Valley are highly restricted to the barrier effect of the mountainous topography of this region, and the fragmentation created by linear developments, fences, and residential & industrial complexes further isolate the herds and limit both seasonal and inter-group migration. The isolation of the Bow Valley elk to such undersized home ranges has the potential to affect social structure, genetic diversity, and density-dependent factors such as disease and reproductive success. In Chapter 4, I investigate the effects of residency in these restricted home ranges on the parasites hosted by elk in the Banff, Canmore, and BVPP herds.

### 3.5 Tables

**Table 3-1** GPS collar deployment and summary of telemetry data. Only three-dimensional fixes were retained.

Collar ID	Deployment Time (days)	Capture Method	Fix Number	Fix Success (%)
<b>Canmore</b>				
1	263	Free Range	5849	92.7
2	231	Free Range	5422	97.8
4	519	Free Range	11827	95.0
32	367	Helicopter	7241	85.2
123	157 <sup>1</sup>	Helicopter	1909	50.7
124	367	Helicopter	8543	97.0
125	367	Helicopter	7789	88.4
6	<i>Collar Lost</i>	<i>Free Range</i>	0	0
<b>BVPP</b>				
22	486	Corral Trap	10303	88.3
25	43	Corral Trap	923	89.4
29	73	Corral Trap	1719	98.1
35/35	201	Helicopter	4708	97.6
40	448	Corral Trap	10198	94.8
43	378	Corral Trap	8579	94.6
39/39	<i>Collar Failure</i>	<i>Helicopter</i>	0	0
45/45	<i>Collar Failure</i>	<i>Helicopter</i>	0	0
<b>Deadman's Flats</b>				
31	367	Helicopter	7763	88.1
36	12	Corral Trap	254	88.2
37	25	Helicopter	563	93.8
39	186	Corral Trap	4320	96.8
103	367	Helicopter	8268	93.9

<sup>1</sup> Total deployment time of 367 days but total collar failure by day 157. Fix success rate was low due to the elimination of all 2d fixes.



**Table 3-2** Bow Valley elk telemetry summary (2008-2011). GPS fix success rate, total telemetry days (green highlights) for each animal, and cause of death are indicated when applicable.

Canmore Collar ID	2008			2009												2010						2011																	
	O	N	D	J	F	M	A	M	J	J	A	S	O	N	D	J	F	M	A	...	D	J	F	M	A	M	J	J	A	S									
1				263 days																																			
2				231 days																																			
4				519 days																																			
32				367 days																																			
123				157 days (collar error)																																			
124				367 days																																			
125				367 days																																			
6				Collar lost (Battery depleted prior to release)																																			

BVPP Collar ID	2008			2009												2010						2011																	
	O	N	D	J	F	M	A	M	J	J	A	S	O	N	D	J	F	M	A	...	D	J	F	M	A	M	J	J	A	S									
22				486 days																																			
25				43 d	Road Mortality (TransCanada)																																		
29				73 days	Road Mortality (TransCanada)																																		
40				448 days																																			
43				378 days												Road Mortality (TransCanada)																							
35/35																																							
39/39																																							
45/45																																							

Deadman's Collar ID	2008			2009												2010											
	O	N	D	J	F	M	A	M	J	J	A	S	O	N	D	J	F	M	A	...	D						
31				367 days																							
36				12	Predation (cougar)																						
37				25 d.	Predation (cougar)																						
39				186 days												Predation (wolf or cougar)											
103				367 days																							

GPS Fix Success Rate	
93-100%	
88-91%	
85-88%	
50%	
No Data	

**Table 3-3** Spatial cohesion of radiocollared elk in the vicinity of the Canmore town site. “n” indicates the number of concurrent fixes in winter/summer, “Total” indicates the percentage of fixes in which animals pairs are correlated (*i.e.* within 200m) on an annual basis, “Winter” indicates percent of fixes animal pairs are correlated (*i.e.* within 200m) during winter (November-April) and median distance, “summer” indicates percent of fixes of animal pairs that are correlated (*i.e.* within 200m) in summer (May-September) and median distance.

Collar ID		<b>1</b>	<b>2</b>	<b>4</b>	<b>32</b>	<b>123</b>
<b>1</b>	<i>n</i>	3311 / 2221	3264 / 2128	19 / 1844	23 / 1228	
	<i>Total</i>	55%	48%	41%	24%	
	<i>winter</i>	<b>54%</b> (162m)	<b>56%</b> (151m)	-	-	
	<i>summer</i>	<b>28%</b> (341m)	<b>22%</b> (1723m)	<b>41%</b> (554m)	<b>24%</b> (2092m)	
<b>2</b>	<i>n</i>		3461 / 1614	19 / 1385	22 / 944	
	<i>Total</i>		46%	45%	35%	
	<i>winter</i>		<b>47%</b> (239m)	-	-	
	<i>summer</i>		<b>26%</b> (1955m)	<b>45%</b> (409m)	<b>35%</b> (1519m)	
<b>4</b>	<i>n</i>			3088 / 2708	23 / 1686	
	<i>Total</i>			63%	15%	
	<i>winter</i>			<b>80%</b> (81m)	-	
	<i>summer</i>			<b>33%</b> (807m)	<b>15%</b> (3429m)	
<b>32</b>	<i>n</i>				13 / 1516	
	<i>Total</i>				20%	
	<i>winter</i>				-	
	<i>summer</i>				<b>20%</b> (2250m)	

Continued Next Page...

Collar ID		124	125
1	<i>n</i>	28 / 2207	31 / 2076
	<i>Total</i>	28%	0%
	<i>winter</i>	-	-
	<i>summer</i>	28% (827m)	0.0% (1222m)
2	<i>n</i>	27 / 1636	29 / 1573
	<i>Total</i>	36%	0%
	<i>winter</i>	-	-
	<i>summer</i>	30% (633m)	0% (7435m)
4	<i>n</i>	3459 / 3241	3287 / 2898
	<i>Total</i>	69%	0.001%
	<i>winter</i>	82% (72m)	0.0% (5807m)
	<i>summer</i>	35% (603m)	0.0% (5035m)
32	<i>n</i>	3101 / 2818	2934 / 2548
	<i>Total</i>	66%	0.001%
	<i>winter</i>	83% (71m)	0.0% (5900m)
	<i>summer</i>	34% (539m)	0.0% (1385m)
123	<i>n</i>	20 / 1760	22 / 1622
	<i>Total</i>	13%	0.001%
	<i>winter</i>	-	-
	<i>summer</i>	12% (2827m)	0.0% (1349m)
124	<i>n</i>		3283 / 3095
	<i>Total</i>		0.01%
	<i>winter</i>		0.0% (5801m)
	<i>summer</i>		0.0% (6793m)

**Table 3-4** Spatial Cohesion of radiocollared elk within Bow Valley Provincial Park. “n” indicates the number of concurrent fixes in winter/summer, “Total” indicates the percentage of fixes in which animals pairs are correlated (*i.e.* within 200m) on an annual basis, “Winter” indicates percent of fixes animal pairs are correlated (*i.e.* within 200m) during winter (November-April) and median distance, “summer” indicated percent of fixes of animal pairs that are correlated (*i.e.* within 200m) in summer (May-September) and median distance.

Collar ID	22	25	29	40	43	35/35
22	<i>n</i>	854 / 0	684 / 867	4949 / 2943	3409 / 2905	0
	<i>Total</i>	<b>72%</b>	<b>37%</b>	<b>50%</b>	<b>58%</b>	-
	<i>winter</i>	72% (78m)	53% (145m)	75% (61m)	70% (88m)	-
	<i>summer</i>	-	25% (1276m)	18% (2137m)	39% (437m)	-
25	<i>n</i>	0	110 / 0	0	0	0
	<i>Total</i>	-	<b>45%</b>	-	-	-
	<i>winter</i>	-	45% (700m)	-	-	-
	<i>summer</i>	-	-	-	-	-
29	<i>n</i>	701 / 919	697 / 916	0	0	0
	<i>Total</i>	<b>53%</b>	<b>41%</b>	-	-	-
	<i>winter</i>	70% (81m)	58% (267m)	-	-	-
	<i>summer</i>	40% (497m)	35% (665m)	-	-	-
40	<i>n</i>	3565 / 3211	0	0	0	0
	<i>Total</i>	<b>59%</b>	-	-	-	-
	<i>winter</i>	70% (83m)	-	-	-	-
	<i>summer</i>	18% (1808m)	-	-	-	-
43	<i>n</i>	0	0	0	0	0
	<i>Total</i>	-	-	-	-	-
	<i>winter</i>	-	-	-	-	-
	<i>summer</i>	-	-	-	-	-

**Table 3-5** Spatial Cohesion of radiocollared elk in the vicinity of Deadman’s Flats. “n” indicates the number of concurrent fixes in winter/summer, “Total” indicates the percentage of fixes in which animals pairs are correlated (*i.e.* within 200m) on an annual basis, “Winter” indicates percent of fixes animal pairs are correlated (*i.e.* within 200m) during winter (November-April) and median distance, “summer” indicated percent of fixes of animal pairs that are correlated (*i.e.* within 200m) in summer (May-September) and median distance.

Collar ID	31	36	37	39	103
31	<i>n</i>	0	0	0 / 2718	3454 / 2621
	<i>Total</i>	-	-	<b>13%</b>	<b>61%</b>
	<i>winter</i>	-	-	-	86% (29m)
	<i>summer</i>	-	-	13% (4684m)	30% (297m)
36	<i>n</i>	0	254 / 0	0	
	<i>Total</i>	-	<b>19%</b>	-	
	<i>winter</i>	-	19% (1126m)	-	
	<i>summer</i>	-	-	-	
37	<i>n</i>		26 / 524	24 / 533	
	<i>Total</i>		<b>23%</b>	<b>0%</b>	
	<i>winter</i>		-	-	
	<i>summer</i>		25% (2085m)	0% (6746m)	
39	<i>n</i>			25 / 3287	
	<i>Total</i>			<b>0%</b>	
	<i>winter</i>			-	
	<i>summer</i>			0% (4823m)	

**Table 3-6** Coefficients for herd level resource selection models of GPS collared elk in the Canmore herd within the Bow Valley region of the Canadian Rocky Mountains.

Habitat Covariate	<i>Canmore (Used)</i>			<i>Canmore (Available)</i>			Data Source	Biological Relevance
	Mean	Min	Max	Mean	Min	Max		
Slope	3	0	47	10	0	66	CEDE <sup>A</sup>	(Sawyer et al. 2007)
Elevation ( <b>Elev</b> )	1339	1292	2361	1461	1290	2447	CEDE	(Sawyer et al. 2007)
Distance to Hydro Feature ( <b>Hydro</b> )	230	0	1276	284	0	1401	NHN <sup>B</sup>	<a href="#">(Johnson et al. 2000)</a>
Distance to Golf Course ( <b>GC</b> )	486	0	6834	1801	0	7324	AVI <sup>C</sup>	Field Observations
Road length within 100m buffer ( <b>Road</b> )	65	0	1004	52	0	1119	NRN <sup>D</sup>	(Rumble et al. 2011) <sup>E</sup>
% Area of Trembling Aspen land cover within 100m Buffer ( <b>V1</b> )	3	0	100	2	0	100	AVI	(Fortin et al. 2005)
% Area of Lodgepole Pine land cover within 100m Buffer ( <b>V2</b> )	6	0	100	14	0	100	AVI	(Hebblewhite et al. 2005)
% Area of White Spruce land cover within 100m Buffer ( <b>V3</b> )	41	0	100	44	0	100	AVI	(Rumble et al. 2011)
% Area of Forb/Grazing land cover within 100m Buffer ( <b>LC1</b> )	38	0	100	14	0	100	AVI	(Hebblewhite et al. 2005)
% Area anthropogenic land cover within 100m Buffer ( <b>LC2</b> )	7	0	100	7	0	100	AVI	(Sawyer et al. 2006)
% Area unsuitable habitat (rock/ice) land cover within 100m Buffer ( <b>LC3</b> )	1	0	100	5	0	100	AVI	(Hebblewhite et al. 2005)
% Area shrub land cover within 100m Buffer ( <b>LC4</b> )	10	0	100	6	0	100	AVI	(Hebblewhite et al. 2005)

A - Canadian Digital Elevation Data (1:50,000 scale Digital Elevation Model - DEM) with 8-23 m grid resolution

B - National Hydro Network (NHN - 1:50,000 scale) dataset with 30m minimum positional accuracy - [www.geobase.ca](http://www.geobase.ca)

C - Alberta Vegetation Inventory (AVI - 1:20,000 scale)

D - National Road Network (NRN) dataset with 10 meter minimum positional accuracy - [www.geobase.ca](http://www.geobase.ca)

E - Rumble et al. 2011 utilized a 250m buffer based on mean nearest neighbour distance, for our study the 100m buffer reflected the median step length between fix locations across all animals

**Table 3-7** Coefficients for herd level resource selection models of GPS collared elk in the Bow Valley provincial park and Deadman's Flats herds within the Bow Valley region of the Canadian Rocky Mountains.

Habitat Covariate	BVPP/Deadman's (Used)			BVPP/Deadman's (Available)			Data Source	Biological Relevance
	Mean	Min	Max	Mean	Min	Max		
Slope	4	0	62	8	0	67	CDED <sup>A</sup>	(Sawyer <i>et al.</i> 2007)
Elevation (Elev)	1350	1290	2555	1379	1238	2184	CDED	(Sawyer <i>et al.</i> 2007)
Distance to Hydro Feature (Hydro)	116	0	1354	408	0	1538	NHN <sup>B</sup>	(Johnson <i>et al.</i> 2000)
Distance to Golf Course (GC)	1910	0	5648	2297	0	6175	AVI <sup>C</sup>	Field Observations
Road length within 100m buffer (Road)	5	0	731	38	0	922	NRN <sup>D</sup>	(Rumble <i>et al.</i> 2011) <sup>E</sup>
% Area of Trembling Aspen land cover within 100m Buffer (V1)	25	0	100	10	0	100	AVI	(Fortin <i>et al.</i> 2005)
% Area of Lodgepole Pine land cover within 100m Buffer (V2)	5	0	100	16	0	100	AVI	(Hebblewhite <i>et al.</i> 2005)
% Area of White Spruce land cover within 100m Buffer (V3)	22	0	100	24	0	100	AVI	(Rumble <i>et al.</i> 2011)
% Area of Forb/Grazing land cover within 100m Buffer (LC1)	12	0	100	28	0	100	AVI	(Hebblewhite <i>et al.</i> 2005)
% Area anthropogenic land cover within 100m Buffer (LC2)	1	0	100	2	0	100	AVI	(Sawyer <i>et al.</i> 2006)
% Area unsuitable habitat (rock/ice) land cover within 100m Buffer (LC3)	2	0	100	4	0	100	AVI	(Hebblewhite <i>et al.</i> 2005)
% Area shrub land cover within 100m Buffer (LC4)	43	0	100	3	0	100	AVI	(Hebblewhite <i>et al.</i> 2005)

**A** - Canadian Digital Elevation Data (1:50,000 scale Digital Elevation Model - DEM) with 8-23 m grid resolution

**B** - National Hydro Network (NHN - 1:50,000 scale) dataset with 30m minimum positional accuracy - [www.geobase.ca](http://www.geobase.ca)

**C** - Alberta Vegetation Inventory (AVI - 1:20,000 scale)

**D** - National Road Network (NRN) dataset with 10 meter minimum positional accuracy - [www.geobase.ca](http://www.geobase.ca)

**E** - Rumble *et al.* 2011 utilized a 250m buffer based on mean nearest neighbor distance, for our study the 100m buffer reflected the median step length between fix locations across all animals

**Table 3-8** Candidate models for elk resource selection functions for the Canmore and Bow Valley Provincial Park regions

Model Number	Parameters	Justification for model design
1	GC	Simple model intended for Canmore specific application predicting strong influence of golf course attraction, all other habitat used proportional to availability
2	LC1	<b>Grazing Selection Model</b> - predicts strong selection based exclusively on grazing landcover, all other habitat used proportional to availability
3	Hydro/LC1	<b>Grazing &amp; Water Selection</b> - predicts strong selection based exclusively on grazing landcover and water availability (used as movement corridor and hydration), all other habitat used proportional to availability
4	GC/Road/LC2	<b>Anthropogenic Model</b> - predicts selection based on all measured anthropogenic effects (distance to golf course, road density, anthropogenic landcover)
5	V1/V2/V3	<b>High Cover Model</b> - predicts habitat selection highly dependent on high cover (coniferous and deciduous) for thermoregulation and predator evasion
6	V1/V2/V3/LC4	<b>Comprehensive Cover Model</b> - predicts selection based strongly on cover including shrubland which comprise major browse contribution
7	V1/V2/V3/LC1/LC4	<b>Comprehensive Vegetation Model</b> - all vegetation landcover classes included in browse/forage/cover centric model. Ignores topographic and anthropogenic effects.
8	Slope/Elev/LC3	<b>Unsuitable Terrain Avoidance Model</b> - Model predicts uniform use of suitable habitat, only unsuitable habitat is avoided (high elevation/steep/alpine rock or ice)
9	Slope/Elev/LC1/LC3	Model predicts topographic effects and habitat choice dictated by grazing selection and avoidance of unsuitable alpine (rock/ice) terrain
10	Slope/Elev/Road/LC1	Model predicts topographic effects, and habitat choice dictated by anthropogenic effects (all areas with roads) and grazing potential
11	Slope/Elev/Hydro/LC1	Model predicts topographic effects, and habitat choice dictated by grazing and water requirements



12	<b>Slope/Elev/Road/V1/V3/LC1</b>	<b>BVPP model</b> - major landcover/features prominent throughout Bow Valley Prov Park.
13	<b>Slope/Elev/Hydro/Road/V3/LC1</b>	Chetkiewicz & Boyce (2009) Model Parameters including primary cover (V3) and grazing (LC1)
14	<b>Slope/Road/V1/V3/LC1/LC4</b>	Anderson (2005) Model Parameters - Roads, Forbs/Grass/Sedges (LC1), Woody Browse (V1, V3, LC4), Slope added as a parameter to account for montane terrain
15	<b>Slope/Elev/Road/V3/LC1/LC2</b>	<b>Canmore Model</b> - prominent landcover/feature classes prominent throughout Canmore range

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**Table 3-9** QIC ranked candidate models for estimating winter and summer resource selection functions in GPS-collared elk within the Canmore town site. Model Parameters include: Slope, Elevation (**Elev**), Distance to Hydro Feature (**Hydro**), Distance to Golf Course (**GC**), Road length within 100m buffer (**Road**), % Area of Trembling Aspen land cover within 100m Buffer (**V1**), % Area of Lodgepole Pine land cover within 100m Buffer (**V2**), % Area of White Spruce land cover within 100m Buffer (**V3**), % Area of Forb/Grazing land cover within 100m Buffer (**LC1**), % Area anthropogenic land cover within 100m Buffer (**LC2**), % Area unsuitable habitat (rock/ice) land cover within 100m Buffer (**LC3**), % Area shrub land cover within 100m Buffer (**LC4**)

RSF Model	<i>Canmore Winter</i>			<i>Canmore Summer</i>		
	QIC	$\Delta$ QIC	Rank	QIC	$\Delta$ QIC	Rank
<b>Slope + Elev + Hydro + Road + V3 + LC1</b>	<b>60288</b>	<b>0</b>	<b>1</b>	<b>39799</b>	<b>0</b>	<b>1</b>
Slope + Elev + Road + V3 + LC1 + LC2	60350	62	2	39972	173	3
Slope + Elev + Road + V1 + V3 + LC1	60427	139	3	39871	72	2
Slope + Elev + Road + LC1	60722	434	4	40354	555	4
Slope + Elev + Hydro + LC1	60871	583	5	41186	1387	5
Slope + Elev + LC1 + LC3	61047	759	6	41279	1480	6
Slope + Road + V1 + V3 + LC1 + LC4	62570	2282	7	41339	1540	7
Slope + Elev + LC3	67312	7024	8	42588	2789	8
V1 + V2 + V3 + LC1 + LC4	69313	9025	9	47708	7909	11
Hydro + LC1	71152	10864	10	48853	9054	12
LC1	71831	11543	11	50005	10206	13
GC + Road + LC2	74204	13916	12	47144	7345	9
GC	74207	13919	13	47606	7807	10
V1 + V2 + V3	79730	19442	14	50851	11052	15
V1 + V2 + V3 + LC4	80251	19963	15	50698	10899	14

**Table 3-10** Coefficients of selection for most supported RSF models for elk in Canmore during winter and summer seasons. Model Parameters include: Slope, Elevation (Elev), Distance to Hydro Feature (Hydro), Road length within 100m buffer (Road), % Area of White Spruce land cover within 100m Buffer (V3), and % Area of Forb/Grazing land cover within 100m Buffer (LC1)

	Code	Parameter	Estimate	S.E. (Empirical)	95% Confidence Limits	
Winter		<i>Intercept</i>	16.2306	4.0938	8.207	24.2543
	Slope	Slope	-0.0636	0.0152	-0.0934	-0.0338
	Elev	Elevation	-0.012	0.0033	-0.0184	-0.0055
	Hydro	Hydro (distance to)	0.001	0.0006	-0.0001	0.0021
	Road	Road Length (100m buffer)	-0.0017	0.0004	-0.0025	-0.0008
	V3	White Spruce (100m buffer)	-0.519	0.0525	-0.622	-0.416
	LC1	Forb/Grazing (100m buffer)	2.297	0.2943	1.7202	2.8738
	Summer		<i>Intercept</i>	19.3822	6.4598	6.7213
Slope		Slope	-0.0701	0.0124	-0.0944	-0.0459
Elev		Elevation	-0.0139	0.0049	-0.0235	-0.0043
Hydro		Hydro (distance to)	0.0009	0.0001	0.0006	0.0012
Road		Road Length (100m buffer)	-0.0035	0.0003	-0.0041	-0.0028
V3		White Spruce (100m buffer)	-0.8229	0.172	-1.16	-0.4858
LC1		Forb/Grazing (100m buffer)	1.4969	0.4517	0.6116	2.3822

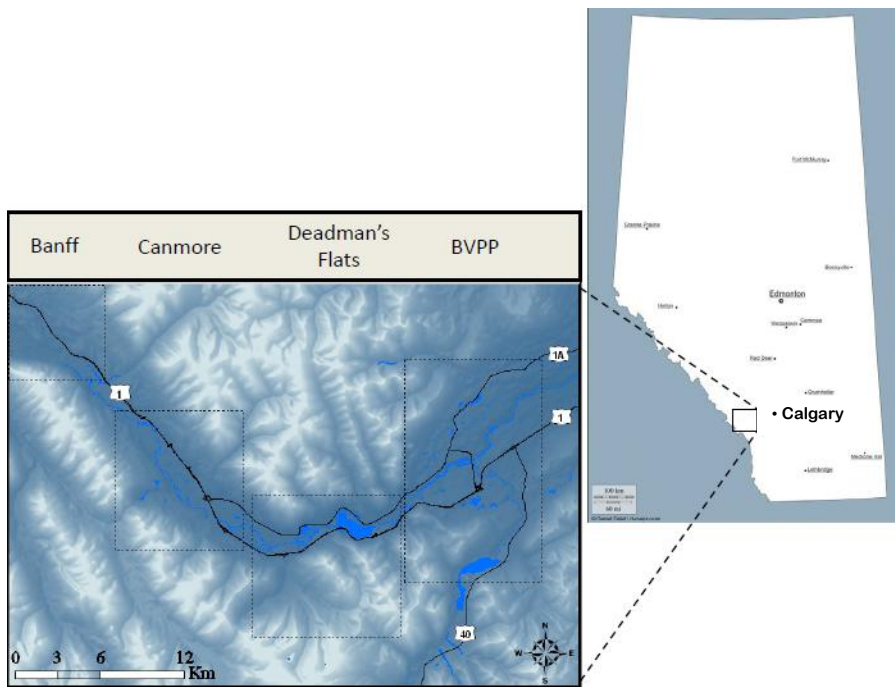
**Table 3-11** QIC ranked candidate models for estimating winter and summer resource selection functions in GPS-collared elk within Bow Valley provincial park and Deadman’s Flats. Model Parameters include: Slope, Elevation (**Elev**), Distance to Hydro Feature (**Hydro**), Distance to Golf Course (**GC**), Road length within 100m buffer (**Road**), % Area of Trembling Aspen land cover within 100m Buffer (**V1**), % Area of Lodgepole Pine land cover within 100m Buffer (**V2**), % Area of White Spruce land cover within 100m Buffer (**V3**), % Area of Forb/Grazing land cover within 100m Buffer (**LC1**), % Area anthropogenic land cover within 100m Buffer (**LC2**), % Area unsuitable habitat (rock/ice) land cover within 100m Buffer (**LC3**), % Area shrub land cover within 100m Buffer (**LC4**)

RSF Models	<i>BVPP Winter</i>			<i>BVPP Summer</i>		
	QIC	ΔQIC	Rank	QIC	ΔQIC	Rank
<i>Slope + Road + V1 + V3 + LC1 + LC4</i>	<b>78765</b>	<b>0</b>	<b>1</b>	<b>44937</b>	<b>0</b>	<b>1</b>
V1 + V2 + V3 + LC1 + LC4	84045	5280	2	47944	3007	2
Slope + Elev + Road + LC1	84416	5650	3	49951	5014	5
Slope + Elev + Road + V1 + V3 + LC1	84493	5728	4	48808	3871	3
Slope + Elev + LC3	84654	5889	5	51115	6178	10
Slope + Elev + LC1 + LC3	84710	5945	6	51004	6067	9
Slope + Elev + Road + V3 + LC1 + LC2	84997	6231	7	50039	5102	7
Slope + Elev + Hydro + Road + V3 + LC1	85010	6244	8	49989	5052	6
Slope + Elev + Hydro + LC1	85469	6703	9	50969	6032	8
V1 + V2 + V3 + LC4	92410	13645	10	48872	3935	4
Hydro + LC1	96252	17486	11	57347	12410	12
V1 + V2 + V3	96947	18182	12	52666	7729	11
LC1	97073	18308	13	58621	13684	14
GC	101033	22268	14	58649	13712	15
GC + Road + LC2	101192	22427	15	58152	13215	13

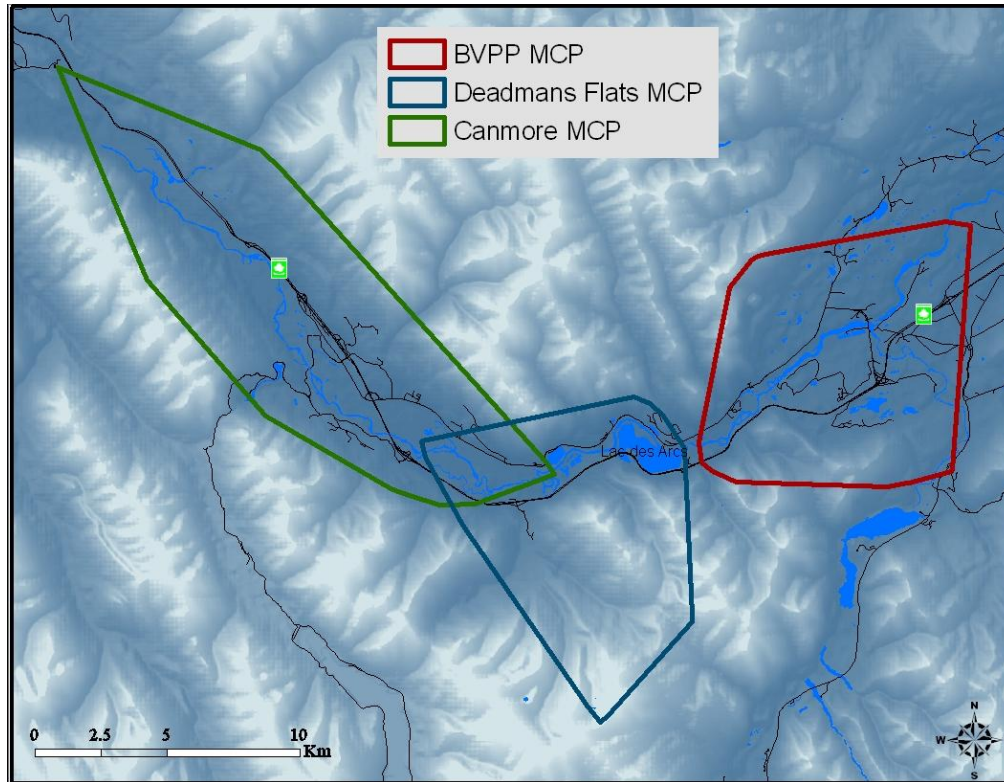
**Table 3-12** Coefficients of selection for most supported RSF models for elk in BVPP/Deadman’s Flats during winter and summer seasons. Model Parameters include: Slope, Road length within 100m buffer (**Road**), % Area of Trembling Aspen land cover within 100m Buffer (**V1**), % Area of White Spruce land cover within 100m Buffer (**V3**), % Area of Forb/Grazing land cover within 100m Buffer (**LC1**), and % Area shrub land cover within 100m Buffer (**LC4**)

	<b>Code</b>	<b>Parameter</b>	<b>Estimate</b>	<b>S.E. (Empirical)</b>	<b>95% Confidence Limits</b>	
Winter		<i>Intercept</i>	-0.6084	0.0838	-0.7727	-0.4441
	Slope	Slope	-0.0956	0.0131	-0.1213	-0.07
	Road	Road Length	-0.0021	0.0003	-0.0028	-0.0015
	V1	Aspen (100m buffer)	0.5605	0.2812	0.0095	1.1116
	V3	White Spruce (100m buffer)	0.9385	0.0973	0.7478	1.1292
	LC1	Forb/Grazing (100m buffer)	1.8461	0.1313	1.5888	2.1033
	LC4	Shrub (100m buffer)	3.2232	0.5461	2.1528	4.2936
	Summer		<i>Intercept</i>	-0.3875	0.1007	-0.5848
Slope		Slope	-0.0886	0.0254	-0.1385	-0.0388
Road		Road Length	-0.0041	0.0005	-0.005	-0.0032
V1		Aspen (100m buffer)	2.1531	0.3356	1.4954	2.8109
V3		White Spruce (100m buffer)	0.9327	0.1639	0.6114	1.2541
LC1		Forb/Grazing (100m buffer)	0.9911	0.1871	0.6243	1.3579
LC4		Shrub (100m buffer)	2.9554	0.5174	1.9413	3.9696

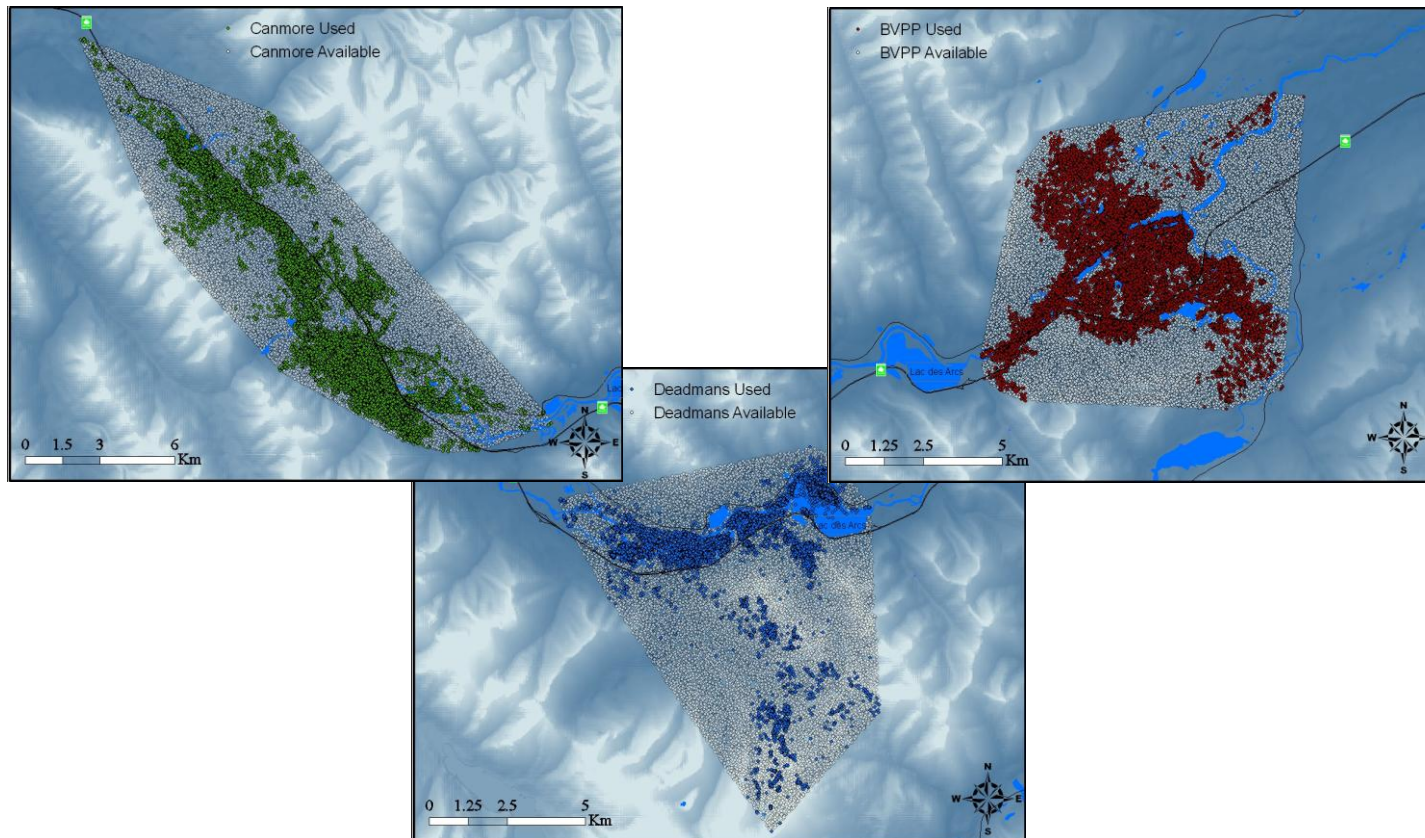
### 3.6 Figures



**Figure 3-1** Overview of the Bow Valley corridor study site in Alberta, Canada. Approximations of regional delineations between herds are shown by dashed boundaries (Banff, Canmore, Deadman's Flats, BVPP). Alberta overview graphic free sourced courtesy [www.d-maps.com/carte.php?num\\_car=23504&lang=en](http://www.d-maps.com/carte.php?num_car=23504&lang=en)

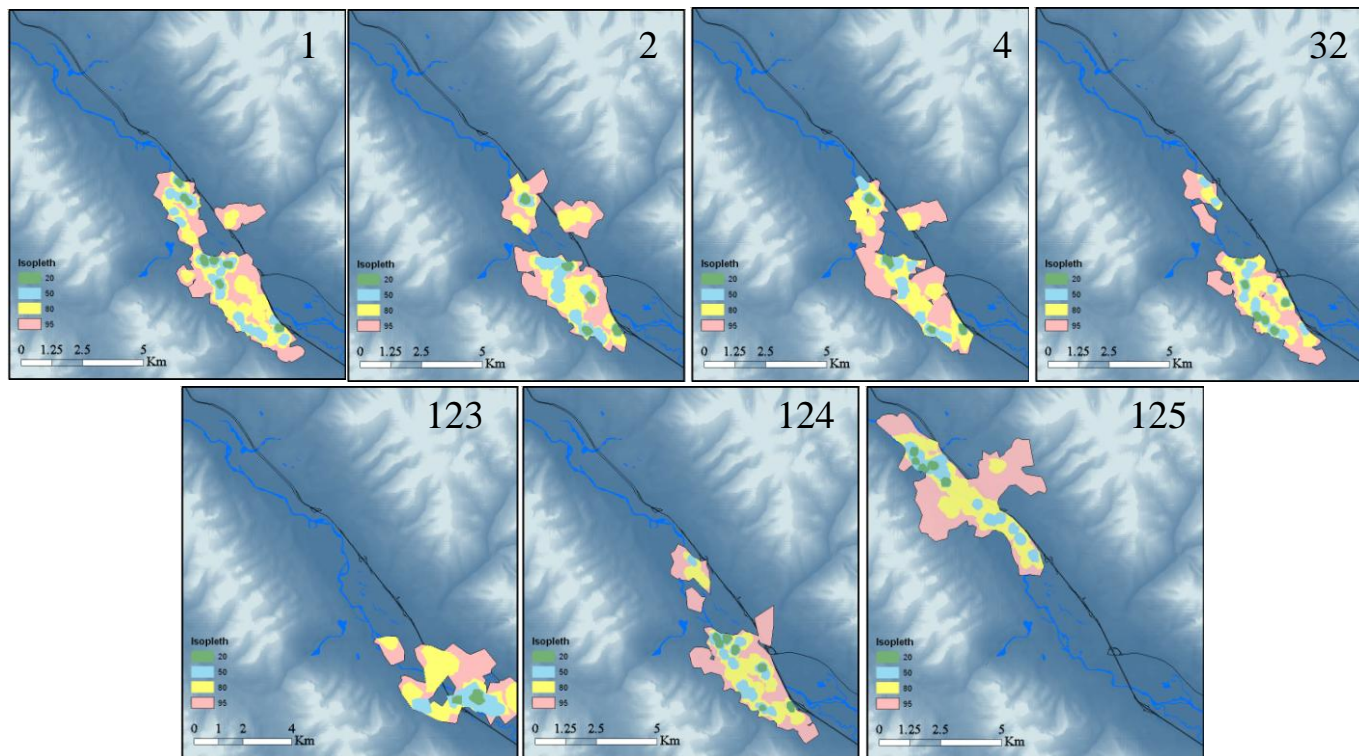


**Figure 3-2** Minimum convex polygon (MCP) home range estimates for Canmore, Deadman’s Flats, and BVPP elk herds. Overlap of Canmore and Deadman’s Flats occurred due to spatial distribution of one collared elk (Canmore 123). The TransCanada highway (Green notation) as well as other primary and secondary roads are displayed, along with the Bow River network. Physical barriers (geologic and anthropogenic) east of Lac des Arcs limits East-West movement within the Bow Valley corridor.

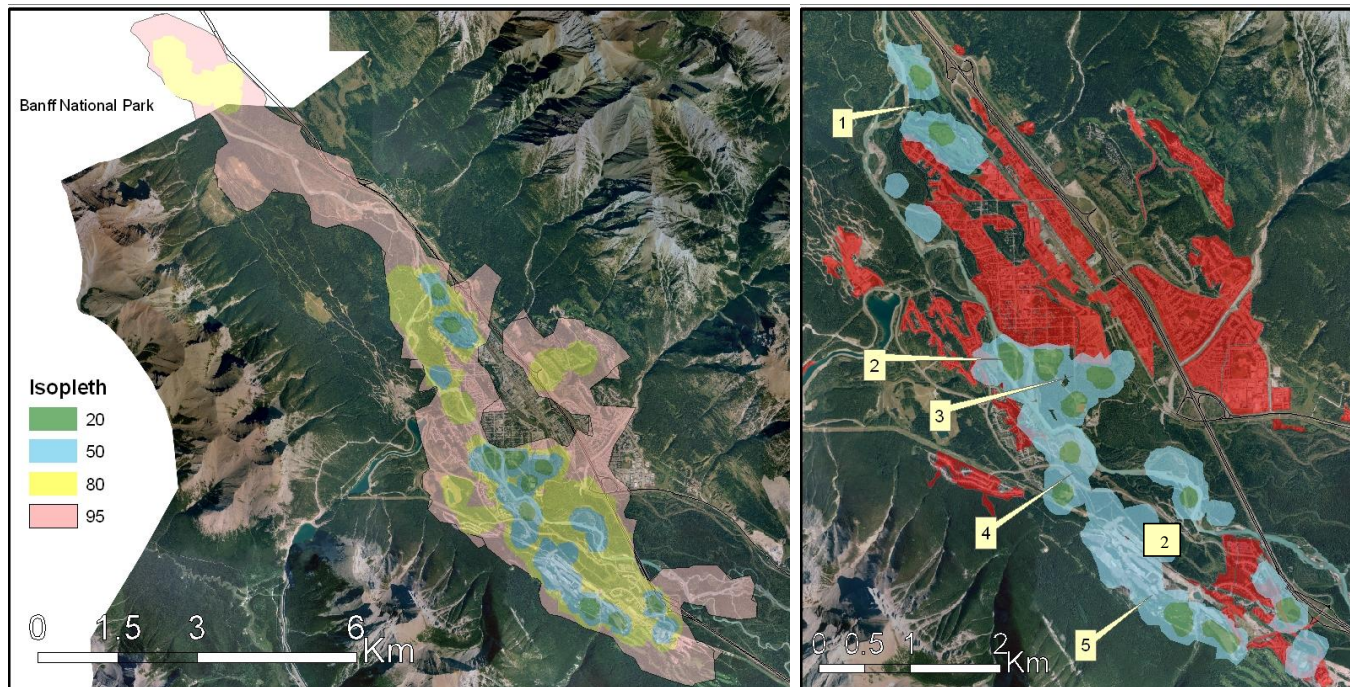


**Figure 3-3** Used (green, red or blue) and available locations (grey) used in the estimation of resource selection function in Canmore, Deadman’s Flats, and Bow Valley Provincial Park herds. Available points were randomly selected within boundaries established via minimum convex polygons encompassing all used telemetry locations.

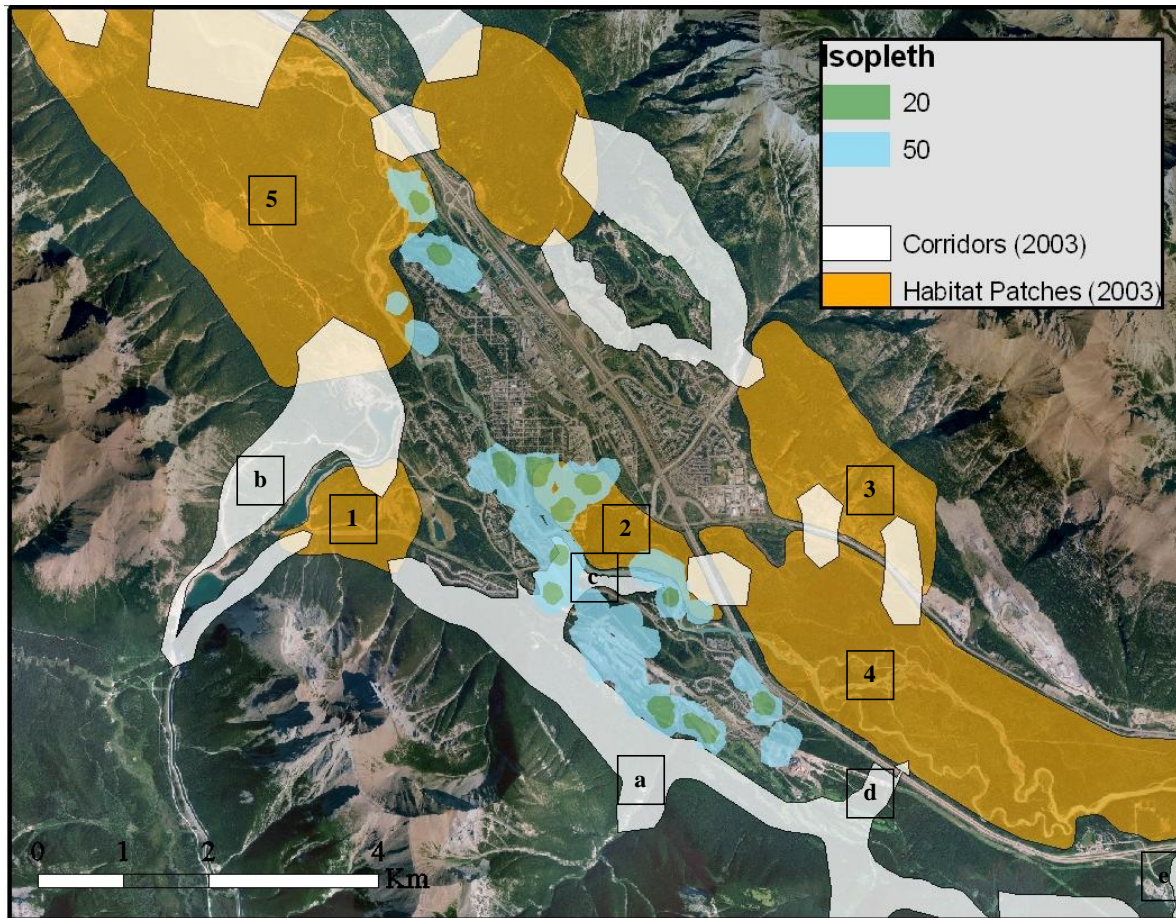




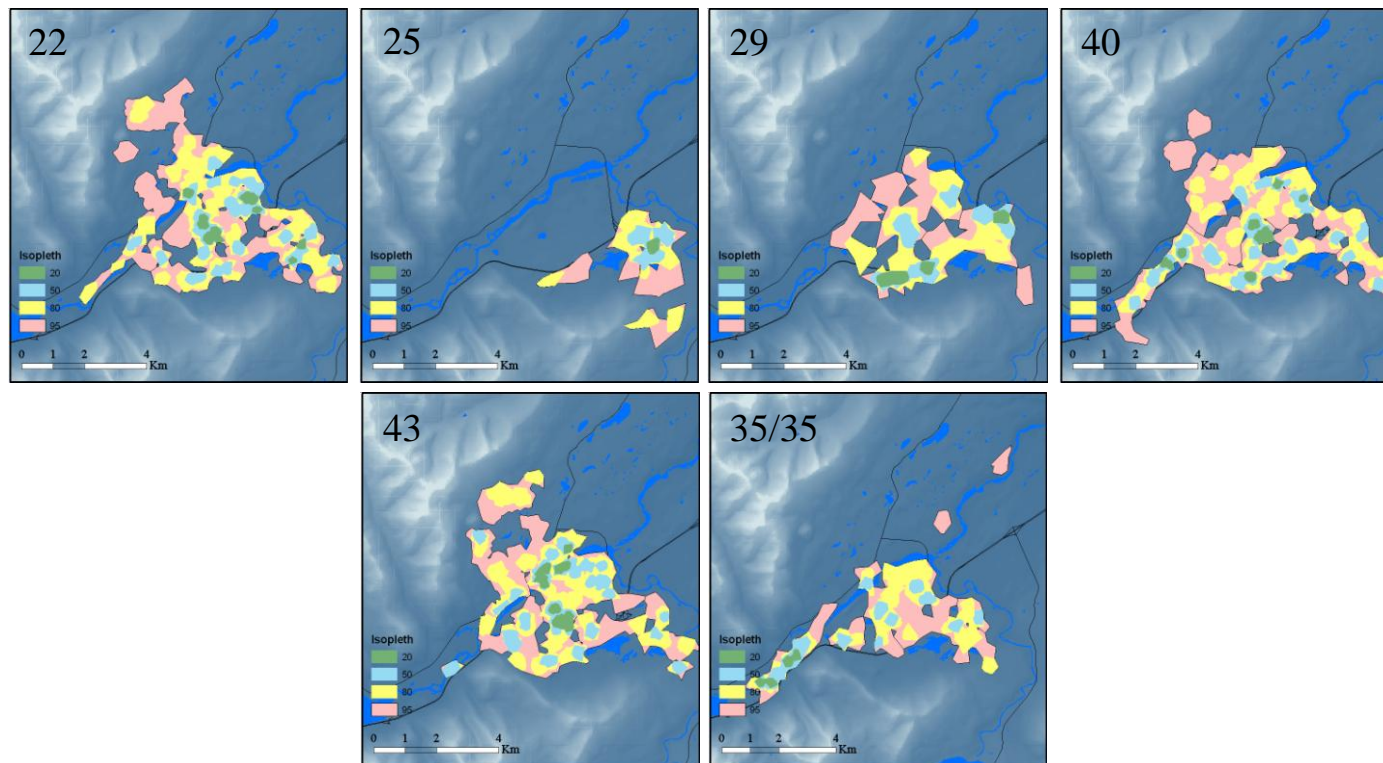
**Figure 3-4** Adaptive local convex hull home range estimates for radiocollared elk within the Canmore town site herd. Individual (1, 2, 4, 32, 123, 124, 125) home range estimates are shown. Utilization distributions within each home range are shown by density of use at isopleths of 20 (green), 50 (light blue), 80 (yellow), and full home range extent is indicated by the 95<sup>th</sup> percentile (pink) isopleth. Areas of highest density of points are in green (20% isopleth).



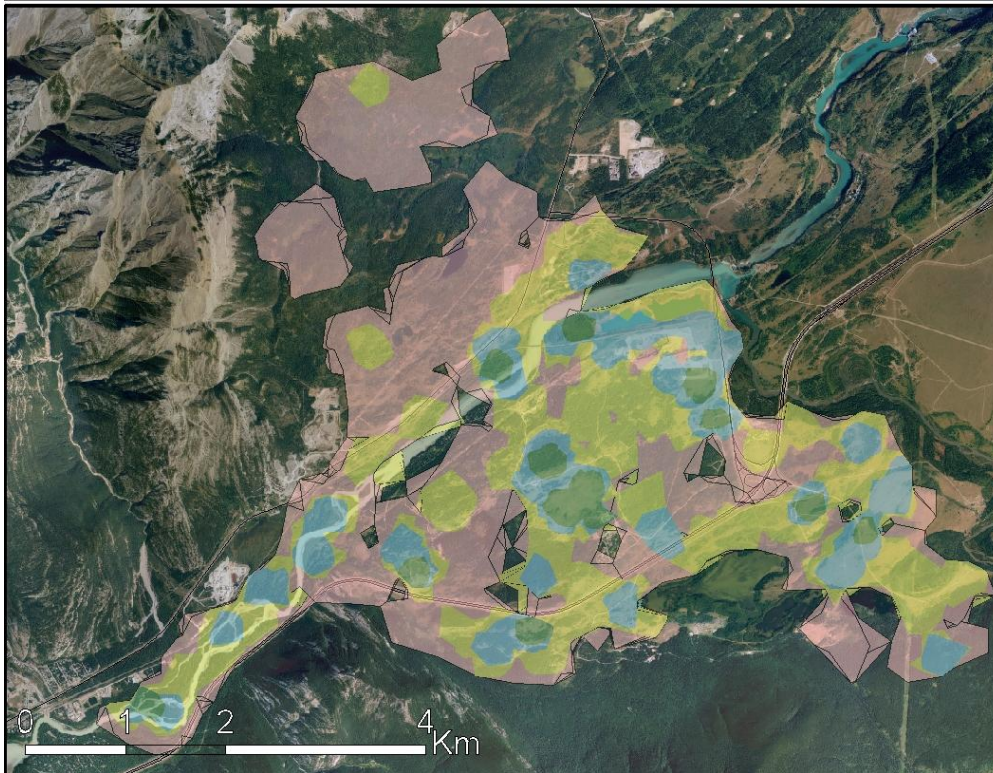
**Figure 3-5** [Left] Final adaptive local convex hull home range for the Canmore elk herd. Utilization distribution isopleths indicate greatest density of use areas by shading color (*i.e.* green isopleth indicates densest 20% portion of telemetry fixes). [Right] “Core” home range areas consisting of 50% of all telemetry fixes. Key habitat features include: (1) Canmore Municipal Golf Course, (2) “Elk Island” – Bow River in-stream island, (3) Soccer fields & municipal parkland, (4) Anthropogenically developed grasslands, and (5) Three Sisters Golf Course development. Commercial, industrial, and residential developments are shaded in red.



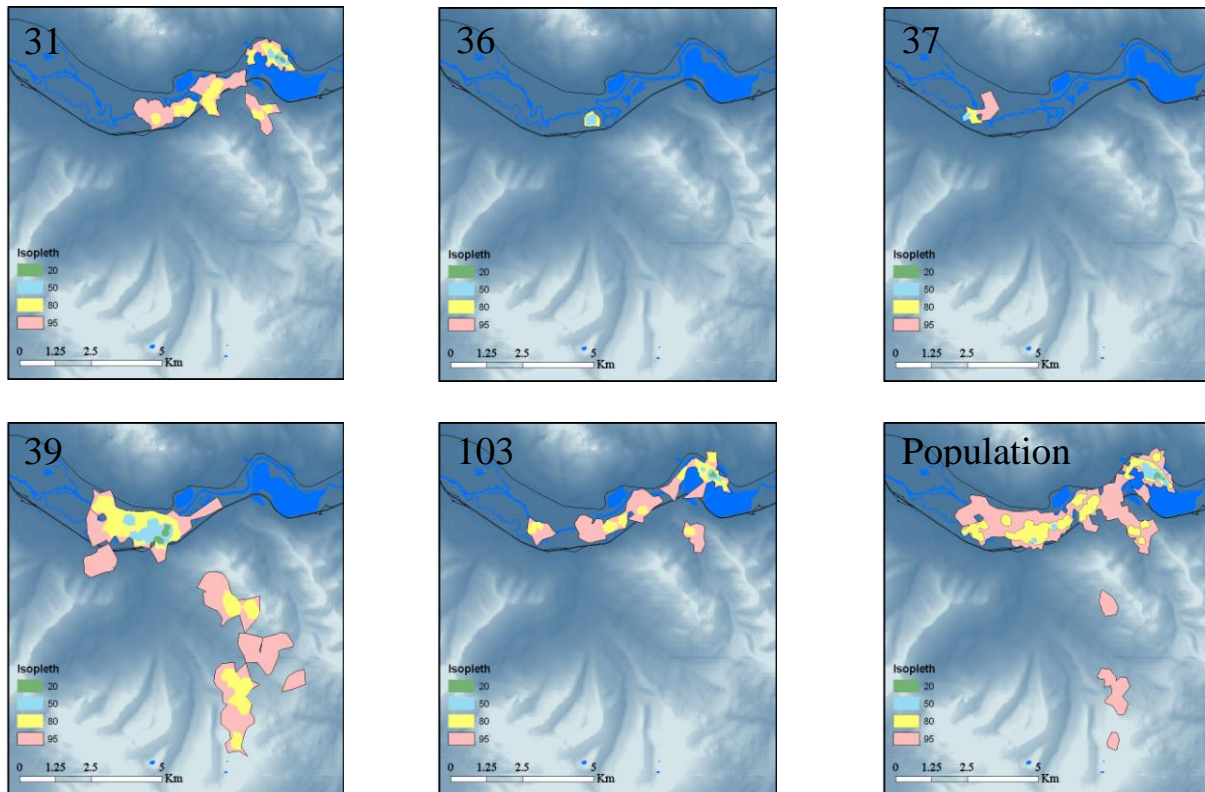
**Figure 3-6** Delineation of important habitat patches and movement corridors in the southern Canmore region as proposed by Alberta Parks, and core home range of the Canmore elk herd. Established habitat patches include: (1) Grassi Lakes, (2) South Canmore, (3) Indian Flats, (4) Bow Flats, (5) Nordic Centre (6) Silvertip; proposed corridors are: (a) Primary multi-species wildlife corridor, (b) Grassi/Canmore Nordic centre corridor, (c) Three Sisters Creek secondary corridor, (d) Stewart primary corridor, and (d) Wind primary corridor.



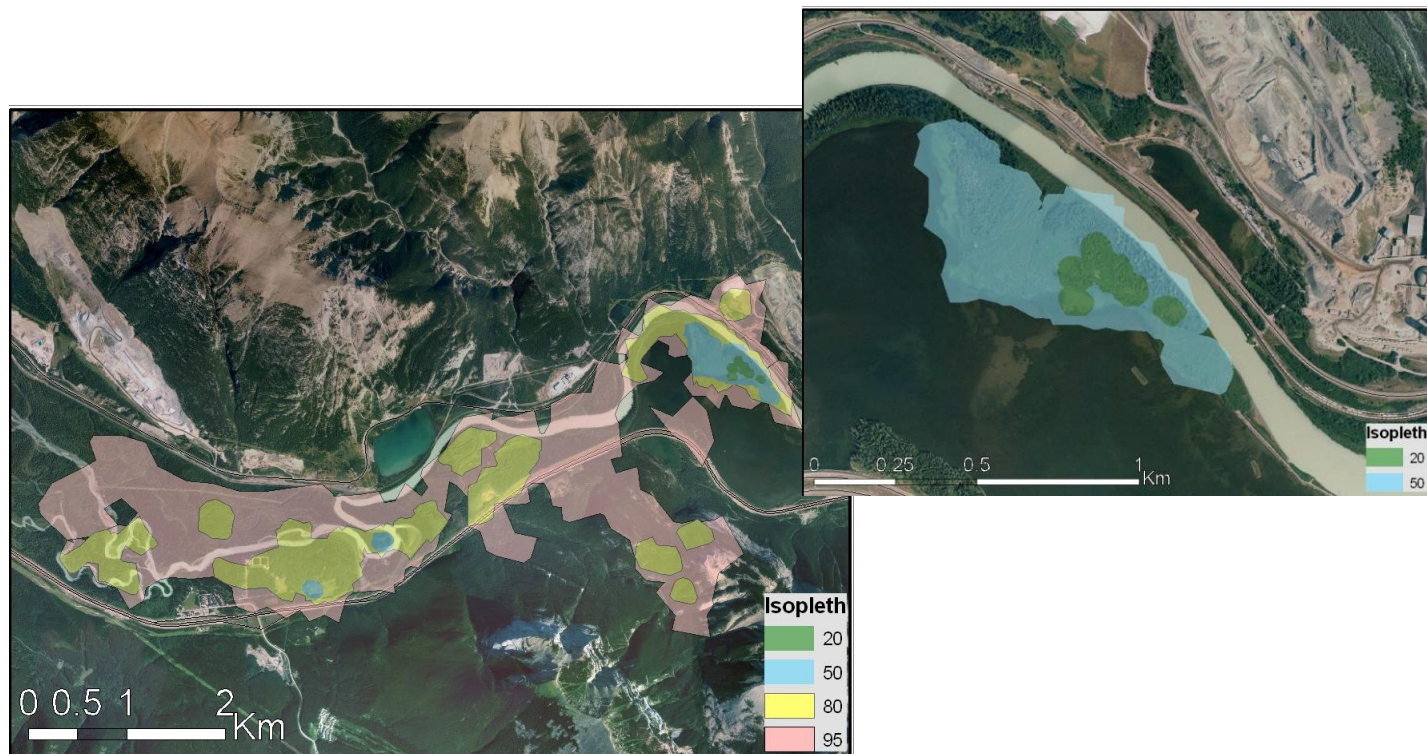
**Figure 3-7** Adaptive local convex hull home range estimates for GPS-collared elk within the Bow Valley Provincial Park herd. Individual (22, 25, 29, 40, 43, 35/35) and herd (pooled telemetry data) home range estimates are shown. Utilization distributions within each home range are shown by density of use at isopleths of 20, 50, and 80 percent; and full extent of home range is indicated by the 95<sup>th</sup> percentile isopleth (*i.e.* encompassing 95% of all telemetry fixes)



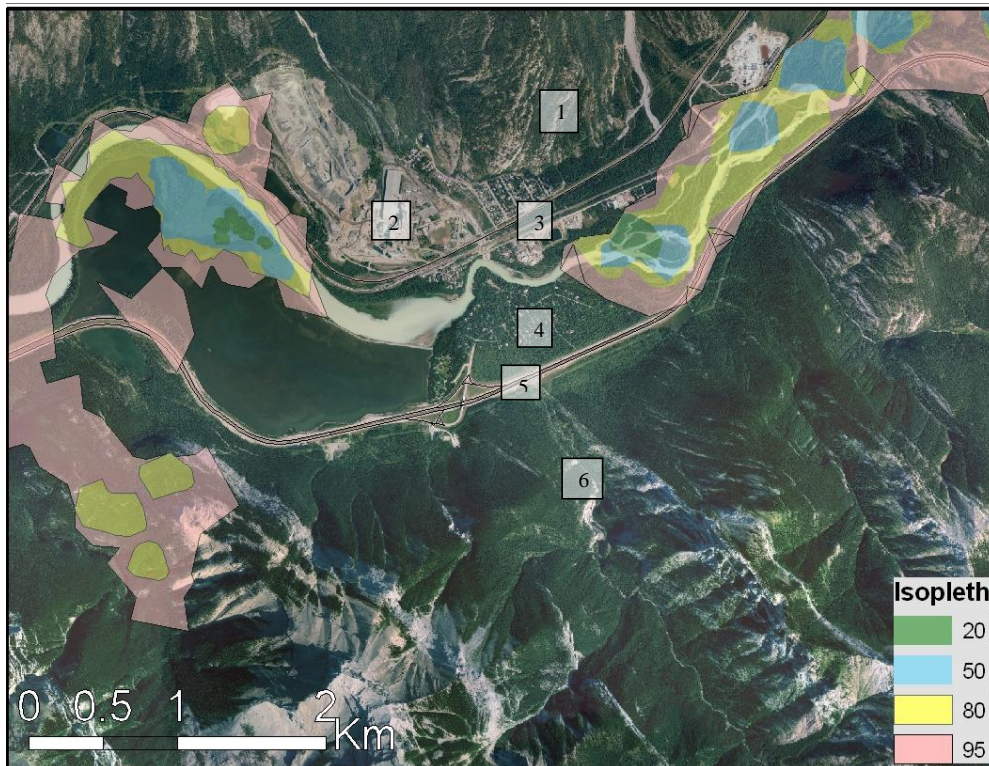
**Figure 3-8** Final adaptive local convex hull home range estimate for GPS-collared elk within the Bow Valley Provincial Park herd. Utilization distribution isopleths indicate greatest density of use areas by shading color (*i.e.* green isopleth indicates densest 20% of telemetry fixes).



**Figure 3-9** Adaptive local convex hull home range estimates for radiocollared elk within the Deadman's Flats herd. Individual (31, 36, 37, 39, 103) and herd home range estimates are shown. Utilization distributions within each home range are shown by density of use at isopleths of 20, 50, 80, and full extent of home range is indicated by the 95<sup>th</sup> percentile isopleth.

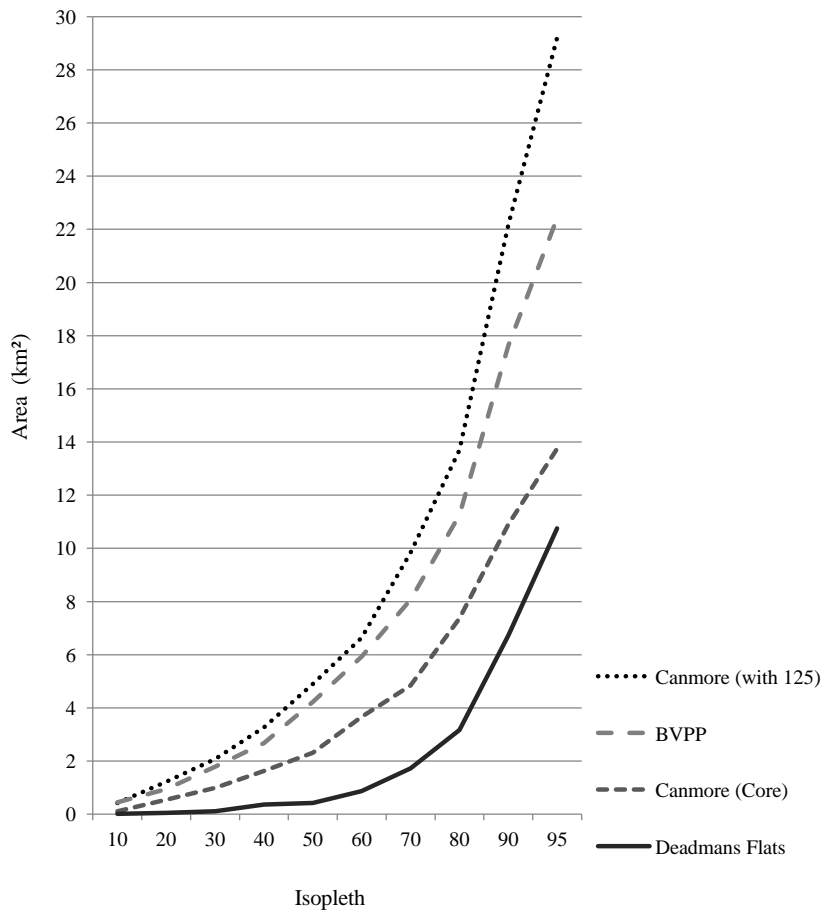


**Figure 3-10** [Left] Final adaptive local convex hull home range estimate for GPS-collared elk within the Deadman's Flats herd. Utilization distribution isopleths indicate greatest density of use areas by shading color (*i.e.* green isopleth indicates densest 20% portion of telemetry fixes). Home range estimates for this herd may be biased due to the mortality of several animals which exhibited higher use of the western portion of the range as well as the Wind Valley (south of TransCanada highway). [Right] Core (50% isopleth) home range at Lac des Arcs.



**Figure 3-11** Detail of physical barriers to East-West dispersion at the intersection of the Deadman's Flats (left) and BVPP (right) elk herds. Barriers include: (1) steep valley wall, (2) Lafarge Exshaw plant, (3) Highway 1a and the Hamlet of Exshaw, (4) Hamlet of Lac des Arcs, (5) Highway 1 (TransCanada), and (6) steep valley wall





**Figure 3-12** Adaptive Local Convex Hull ( $\alpha$ -LoCoH) home range Utilization Distribution isopleth areas, for pooled collar data from cow elk in Bow Valley Provincial Park, Deadman’s Flats and Canmore (home ranges are presented for “core” Canmore resident animals - excluding habitat use by elk 125 which utilized non-urban terrain in Banff National Park, and cumulative home range including elk 125 in the pooled data).

## **4 Host parasite assemblages of resident elk (*Cervus elaphus*) herds within the Bow Valley**

### **4.1 Introduction**

Parasites play a critical role in the regulation of host population size and demography with a magnitude equal to the effects of predation pressure or resource limitation (Nunn, Altizer et al. 2003; Puttker, Meyer-Lucht et al. 2008). High-intensity parasitic infections may lead to reductions in reproductive potential due to altered host physiology (Kristan and Hammond 2000), and the avoidance of infected individuals as potential mates (Ehman and Scott 2002; Kavaliers, Choleris et al. 2005). High-intensity infections may also increase an organisms susceptibility to predation (Murray, Cary et al. 1997).

Factors such as climate change (Kutz, Hoberg et al. 2004), habitat fragmentation (Puttker, Meyer-Lucht et al. 2008; Mbora and McPeck 2009), or anthropogenic habitat disturbance (Patz, Olson et al. 2008) can modify ecosystems and concentrate individuals within restricted or spatially shifted ranges which promotes the transmission of parasites (Anderson and May 1978). Contributing health effects of environmental contaminants or physiological stress from habitat destruction can lead to suppression of immune responses in these already challenged animals (Holmes 1996), compounding the disruptive effects on naturally occurring host-parasite equilibria. This loss of “natural” host-parasite interactions can affect host-parasite dynamics in several ways, including: host ranging, host density, inter- and intraspecific contacts and diet; which may cumulatively affect the ability of hosts to survive parasitic challenges in habitat fragments (Gillespie and Chapman 2008).

As parasite pathogenicity is often directly correlated with parasite numbers and their ability to cause disease (Dobson and May 1986), factors that increase close contact (such as high animal densities) and promote transmission (*i.e.* crowding of animals on preferred areas), are also likely to directly and indirectly increase the degree of pathogenicity and the magnitude of parasite effects on host populations (Holmes 1996; Hoberg, Kocan et al. 2008). Host density is considered to be the greatest parameter in determining infection rates in directly transmitted species, showing strong positive correlation with parasite prevalence and diversity (Gillespie and Chapman 2008).

When pathogens achieve these high densities within reservoir populations, spillover into secondary or aberrant hosts may also occur (Power and Mitchell 2004). This spillover has the potential to shape population and community structure; in particular the transmission of parasites originating from domesticated animals, which could have drastic effects on wild populations (Holmes 1996). Increased urbanization of wild canids such as coyote (*Canis latrans*) has presented several public health concerns due to amplifications of endemic *Echinococcus multilocularis* infections in wild populations persisting within municipalities, and the potential for an anthropogenic life cycle of the parasite involving domestic dogs and cats; along with humans, as final hosts (Catalano S., Lejeune M. et al. 2012)

Host-parasite assemblages can be sensitive indicators of human mediated ecological transformations, such as climate change or habitat disruption (Hoberg, Polley et al. 2008; Kutz, Jenkins et al. 2009) which result in altered ranging of parasites or their hosts, shifting inter/intraspecific associations, altered transmission of endemic parasites, host switching (Hoberg, Kutz et al. 2002), disease emergence (Dobson, Kutz et al. 2003;

Davidson, Simard et al. 2011) or land-use changes (Kutz, Jenkins et al. 2009). In order for such investigations to serve as indicators, baseline investigations on host-parasite assemblages must occur (Hoberg, Polley et al. 2008). A detailed survey of parasite fauna is a critical component to recognizing altered host-parasite dynamics, emergence of new pathogens, and interactions between domestic and sylvatic host populations and should be an integral component in the assessment of biodiversity at local, regional, or global scales (Brooks and Hoberg 2000; Hoberg, Kocan et al. 2008).

Despite the importance of parasitic disease as emerging threats in fragmented ecosystems, relatively few studies have examined how anthropogenic disturbance affects parasite burden within parasite-host assemblages (Murray, Cary et al. 1997; Mbora and McPeck 2009). In this thesis, I aim to provide a baseline measurement of host-parasite assemblages within elk (*Cervus elaphus*) resident to montane habitats along the Bow Valley corridor in the front ranges of the Canadian Rocky Mountains.

Excluding viral and bacterial pathogens, there are approximately 54 species of parasites (See Table 4-1) consistently reported in elk within North America (Toweill and Thomas 2002). Excluding ectoparasites such as ticks and mites, transmission of many of these pathogens is direct via ingestion of infective eggs or larvae through fecal contamination of feeding grounds. This exposes grazers to the risk of infection or re-infection during feeding; especially so if the animals exploit particular grazing locations intensively and repeatedly (Grenfell 1992; Mbora and McPeck 2009). Parasitic infestations are typically low intensity infections of one or multiple parasites, manifesting asymptotically or with mild symptoms such as slight reductions in appetite or incomplete nutrient assimilation (Smits 1991; Holmes 1996; Hoberg, Kocan et al. 2008).

High intensity infections of native parasites due to habitat alteration (Patz, Graczyk et al. 2000; Mbora and McPeck 2009), or opportunistic infection by exotic species introduced through translocations or immigrations (Wasel, Samuel et al. 2003; Jacques and Jenks 2004; deBruyn 2010), may result in significant mortality in wild populations directly due to pathological effects or indirectly through negative effects on host condition causing increased susceptibility to predation or weather-related natural mortality (Corn and Nettles 2001; Gillespie and Chapman 2006).

A primary goal of this investigation is to provide a baseline catalogue of species present in elk within this range; including potentially zoonotic species capable of host switching between domestic and wild animals. This study will also help to determine how the relative degree of anthropogenic disturbance, such as habitat fragmentation and the creation of artificial green spaces can affect the diversity and prevalence of parasitic organisms. I hypothesized that the increased fragmentation of disturbed habitats compounded with attractive habitat classes are resulting in recurrent use of a restricted number of preferred foraging patches by animals close to or within municipal centers, thus increasing animal density at these sites and increasing opportunities of transmission and infection. Due to the fact that fragmentation limits foraging and travel routes, high density populations will often exploit the same habitat patches intensively, which results in high environmental contamination with fecal matter and thus infective eggs and larvae (Mbora and McPeck 2009), which in turn increase the probability that an infective stage will reach a host. This is predicted to result in increased intensity of infection and greater parasite diversity within the Banff and Canmore animals in relation to wild ranging communities such as the Bow Valley Provincial Park herd.

## 4.2 Methods

### Study Site

See Section 3.2 for an overview of the Bow Valley region.

### Sample Collection and Storage

Fecal egg counts (FEC) are a common non-invasive means to estimate parasite intensity and abundance and identify diversity based on the appearance of parasite propagules (oocysts, eggs, or larvae) in the host feces (Ortiz, Ybanez et al. 2000; Villanua, Perez-Rodriguez et al. 2006). FEC are estimates of infection intensities rather than absolute values, as egg counts may be subject to great variability due to host reproductive status (Ruiz de Ybanez, Goyena et al. 2004), season (Kumba, Katjivena et al. 2003), diet (Ezenwa 2004), ingesta volume, and host resistance (Tarazona 1986). Several studies on captive populations of red deer (Lancaster and Andrews 1991; Goossens, Vercruysse et al. 2005), North American elk (Goossens, Vercruysse et al. 2005), and mohor gazelle (*Gazella dama mhorr*) (Ruiz de Ybanez, Goyena et al. 2004), have identified peaks in fecal egg output during the periparturent period. Both wild and captive North American elk and European red deer exhibit lower FEC during the winter months (November-March), with peaks during spring (early April-May) and coinciding with the rut (September-October) (Goossens, Vercruysse et al. 2005; Hines, Ezenwa et al. 2007).

Three herds were targeted for sampling: Bow Valley Provincial Park, Town of Canmore, and the Town of Banff. In order to maximize the probability of detecting potential parasitic infection through egg recovery, sampling efforts coincided with typical

peaks in fecal egg output. Over two successive years of sampling (2010 & 2011), sampling bouts occurred in early April (pre-parturition), early June (post-parturition), and October (coinciding with the rut). Sampling bouts were completed across all three locations within a span of ten days such that temporal effects due to collection dates were minimized and egg counts could be compared between locations by season. Estimates from aerial surveys by the national and provincial parks organizations and complete herd counts during collections place animal density highest in Banff (>200 animals – Hebblewhite, 2000), second-highest in Canmore (70-80), and lowest in Bow Valley Provincial Park (<70 animals within a home range larger than Canmore). Animal identification was limited to ear-tagged individuals, thus the majority of fecal samples could not be accurately associated with a specific animal between sampling bouts to allow a proper longitudinal study. Pooled fecal sampling is possible when assessing egg shedding in ungulates (Ortiz, Ybanez et al. 2000), so for each sampling bout a random sample of animals was selected and I analyzed host-parasite assemblages at a herd-level resolution.

Fecal sampling was accomplished by tracking groups of animals to bedding locations where pellets were deposited. Visual and photographic tracking allowed individual pellet groups to be distinguished for collection once the animals vacated the location. Repeat sampling of individuals within bouts was avoided and only a single fecal sample was taken from each adult during pellet collection.

Great variability in FEC between individuals necessitated that a minimum of 15-20 individuals be sampled per herd to obtain an accurate assessment of the parasite burden of that herd (Gasbarre, Smith et al. 2009), and each fecal sample consisted of

~20g of fresh pellets to ensure sufficient material for parasite extraction (Welch, Pybus et al. 1991). Only fresh pellets were collected to avoid losses due to egg hatching and/or larval migration (Ballweber 2006). All samples recovered were stored at -20 degrees Celsius as soon as possible after collection. Preservation of fecal samples through freezing has been reported to diminish the recovery of eggs and larvae (Foreyt 1986) with the limit of acceptable losses occurring after roughly three months of storage at -20 Celsius (deBruyn 2010). As a precaution, all laboratory protocols for egg and larvae recovery were completed within 12 weeks of the collection date.

### **Laboratory Protocols**

#### *Wisconsin Double Centrifugation Technique (See Appendix B - Part 1)*

Gastrointestinal strongyles (See Appendix A: Figure 1) are recovered using a modified Wisconsin double centrifugation protocol as proposed by Egwang and Slocombe (1982) using a sucrose (Sheather's) solution. The higher specific gravity of the Sheather's solution will cause eggs to rise during centrifugation.

#### *FlukeFinder (See Appendix B – Part 2)*

Flukefinder extractions were used to detect eggs of liver flukes (*Fasciola sp.*, *Fascioloides magna*) that will not float using the modified Wisconsin technique and Sheathers solution.

#### *Fecal egg counts (FEC)*

Slides prepared with the Wisconsin double centrifugation technique were examined at 100x magnification under a compound microscope. FlukeFinder assays were examined under a dissecting microscope. Multiple grams of feces are required for each



assay, but standard final egg counts are reported as eggs per gram of feces (with egg counts rounded up to whole numbers), and parasite identification was as precise as possible from egg morphology alone with the assistance of Dr. S. Kutz and her lab.

### **Data Analysis**

Preliminary descriptive statistics (*i.e.* mean prevalence, mean intensity) were calculated using pooled 2010 and 2011 samples by location (Banff, Canmore, & BVPP) in order to identify major trends in host-parasite assemblages within the sampled herds. The coarse resolution and small sample size provided by our sampling regime was unlikely to discern short term or small scale effects through micro-analysis on an individual or seasonal basis.

Processes provided through the statistical package *Quantitative Parasitology 3.0* (Appendix B – QP 3.0) (Rozsa, Reiczigel et al. 2000) was used to calculate adjusted Wald 95% confidence intervals for parasite prevalence (proportion of individuals infected) across Banff, Canmore, and BVPP herds. I used Fishers exact test; provided in the QP 3.0 framework (Rozsa, Reiczigel et al. 2000), with Dunn-Šidák corrected  $\alpha$  (Abdi 2007) for pair wise comparisons of prevalence between study sites (Abdi 2007).

I calculated bias-corrected and accelerated (BCa) bootstrapped confidence intervals for mean intensity (mean eggs per gram in infected individuals, number of bootstrap replications = 2000) of parasitic infections for each species by herd using tools in the QP 3.0 package (Rozsa, Reiczigel et al. 2000). Mean intensities between herds were compared using bootstrapped 2-sample t-tests (2000 bootstrap replications). In order to account for inflation resulting from multiple comparisons, and to reduce the

likelihood of type-1 errors, Dunn-Šidák corrections for multiple comparisons were used to select a more stringent bounds for significance testing (Abdi 2007).

*Fascioloides magna* eggs had greater potential for recovery and detection (due to size) than the substantially smaller gastrointestinal parasite eggs. Thus, egg counts were more likely to be representative of actual host-parasite assemblages as fewer eggs were likely to be missed during counts. To determine the effects of season and location on parasite intensity, generalized linear models with negative binomial distributions (SAS 9.3 PROC GENMOD); to account for overdispersion and zero-clustering in parasite egg count data (Wilson and Grenfell 1997; Alexander 2012), were fitted to *Fascioloides magna* egg count data separately by collection year (2010 & 2011). Model parameters included: sampling location, season, and the interaction of location\*season. AIC scores were used to determine model fit. Based upon the most supported model, differences of least squares means between locations were compared (within seasons) - (SAS 9.3 PROC GENMOD LSMEANS DIFF). In order to account for the inflation resulting from multiple comparison testing and to reduce the likelihood of type-1 errors, Dunn-Šidák corrections for multiple comparisons were used to select a more stringent significance level (Abdi 2007). Logit scale LSMEANS model estimates were backtransformed to the scale of the original data by applying the inverse link function (SAS 9.3 LSMEANS iLINK) for the purposes of data plotting.

Parasite diversity was fitted separately by year (2010 & 2011) using a generalized linear model (SAS 9.3 PROC GENMOD) with a normal distribution (Wilson and Grenfell 1997; Alexander 2012). Model parameters included: sampling location, season, and the interaction of location\*season. AIC scores were used to determine model fit.

Based upon the most supported model, differences of least squares means between locations were compared (within seasons) - (SAS 9.3 PROC GENMOD LSMEANS DIFF). In order to account for the inflation resulting from multiple comparison testing; and to reduce the likelihood of type-1 errors, Dunn-Šidák corrections for multiple comparisons were used to select a more stringent significance level (Abdi 2007).

### 4.3 Results

#### Prevalence & Mean Intensity

Fecal egg counts identified a number of gastrointestinal (GI) nematodes and one fluke (*Fascioloides magna*) in fecal samples from sampled herds (Table 4.2). Two distinct morphotypes of *Eimeria sp.* were detected sporadically in samples from all three locations, but detectability was determined to be too low to make reliable estimates of prevalence (proportion of animals infected within a herd) or intensity (average number of parasites per infected individual). All locations were thus determined to be positive for both morphotypes of *Eimeria* but prevalence/intensity remains undetermined.

For *F. magna* as well as the GI nematodes, prevalence was typically highest in Banff and Canmore relative to BVPP (no significant difference between Banff and Canmore), while mean intensity (average EPG in infected individuals) for common parasite species (*Fascioloides magna*, *Capillaria sp.*, *Trichostrongylus sp.* "A") was significantly higher in Banff and Canmore relative to BVPP, and occasionally highest in Banff relative to Canmore (Table 4.3). Several rare or poorly detected species (*Trichostrongylus sp.* "B", *Nematodirus sp.* & *Trichuris sp.*) were extremely diffuse across all herds and showed no specific pattern in regards to distribution. (Table 4.2)

Prevalence of *F. magna* in Banff was significantly higher than BVPP, significantly higher in Canmore relative to BVPP; but did not differ significantly between Banff and Canmore (Table 4.2 & 4.3). Mean intensity was significantly higher in Banff than BVPP, higher in Canmore than BVPP; and higher in Banff relative to Canmore (Table 4.2 & 4.3).

*Capillaria sp.* was only detected in Banff and Canmore herds and there was no significant difference in prevalence between Banff and Canmore, while mean intensity was significantly higher in Banff (Table 4.2 & 4.3).

Prevalence of *Trichostrongylus sp. "A"* was significantly higher in Banff relative to BVPP, higher in Canmore than BVPP; and did not differ between Banff and Canmore (Table 4.2 & 4.3). Mean intensity was significantly higher in Banff than BVPP, higher in Canmore and again did not differ between Banff and Canmore (Table 4.2 & 4.3).

A smaller morphotype of *Trichostrongylus sp. "B"* had higher prevalence in Banff than BVPP, higher in Canmore relative to BVPP, and no difference between Banff and Canmore (Table 4.2 & 4.3). Mean intensity did not differ between Banff and BVPP, Canmore and BVPP, or Banff and Canmore (Table 4.2 & 4.3).

Prevalence of *Trichuris sp.* was significantly higher in Banff relative to BVPP, and in Banff relative to Canmore, but was not significantly different between Canmore and BVPP (Table 4.2 & 4.3). Mean intensity was significantly higher in Banff relative to Canmore, and did not differ between Canmore and BVPP, or Banff and BVPP (Table 4.2 & 4.3).

*Nematodirus sp.* was detected sporadically in low prevalence (0.9-3%) across the Banff Canmore and BVPP herds with no significant differences between locations. Mean intensities were unlikely to be accurate due to the small number of positive samples.

#### ***Fascioloides magna* & overall diversity**

Negative binomial GLM models were run stepwise from the full factorial model and ranked based on AIC scores (See Table 4.4). The full model (season + location + location\*season) was the top ranked model in each case with the exception of *F. Magna* FEC in 2010.

Comparisons of least squares means between sampling locations were subject to Dunn-Šidák corrections whereby comparisons for *F.magna* (2010)  $\alpha=0.017$ , and all other comparisons  $\alpha=0.0057$ . General trends suggested greatest fecal egg counts and diversity in Banff, while both Canmore and Banff showed increasing trends in fecal egg counts and diversity as season's progressed (spring through fall).

Comparisons of least squares means for *F.magna* in 2010 indicate no significant effect of season and no significant difference in estimated FEC between Canmore and BVPP , while Banff had significantly higher FEC than Canmore (Figure 4.2).

*F. magna* fecal egg counts in 2011 were significantly affected by season, location, and the interaction thereof. There was no significant difference between mean FEC estimates of Canmore and Banff in spring, summer, or fall; Canmore and Banff both had significantly higher *F.magna* FECs than BVPP during both the spring and the fall; while summer counts were similar across all three locations (Figure 4.3)

Overall, BVPP had low fecal egg counts and low species diversity and infected individuals were often host to assemblages of a single detectable parasite (Figure 4.4 & 4.5) In 2010, parasite species diversity was similar between Banff and Canmore during spring, while higher in Banff during summer and fall seasons; Banff samples displayed greater parasite diversity than BVPP during all seasons; and diversity in Canmore was comparable to BVPP during spring and summer, but greater during the fall ( $Z=3.89$ ,  $p=0.0001$ ). (Figure 4.4)

In 2011, species diversity in Banff and Canmore was similar across all three seasons, and were each significantly greater than BVPP in all three seasons (Figure 4.5).

#### 4.4 Discussion

The high prevalence of several parasitic species within fragmented, urban animals suggests that parasites have the potential to play an important role in herd dynamics throughout the Bow Valley. Of all the parasitic species collected during this survey, *F. magna* is likely of greatest concern. High intensity fluke infections can result in significant liver damage and mortality in infected wildlife, and these infections have the potential to cross over to domestic species sympatric with infected wildlife; which is a distinct possibility in the Eastern portions of the study area where elk habitat begins to intersect with farms and ranchlands (Kingscote, Yates et al. 1987; Pybus 1990; Kennedy, Acorn et al. 1999). Established reservoirs of liver flukes were first identified within the Bow Valley (Banff & Kootenay National Parks) between 1984 and 1989, when they were detected at high prevalence in elk (86%) and associated with extensive damage to liver tissue in infected animals (Pybus, 1990). Alterations to patterns of transmission of these endemic parasites via human mediated habitat disruption could have the potential to increase health effects and animal mortality rates (Dobson, Kutz et al. 2003; Davidson, Simard et al. 2011).

*Nematodirus sp.* has been associated with mortality resulting from gastric enteritis in farmed elk (Woodbury, Berezowski et al. 2005), although at current levels of infection rates and intensities within the Bow Valley, it is unlikely to pose a significant threat. Demographic crashes in high-density captive populations of roe deer have been linked to infection intensities measured via gastrointestinal parasite fecal egg counts in excess of 500 eggs/g feces (Maublanc, Bideau et al. 2009) which is well above the estimates of this study; although our results are likely to underestimate true infection rates as a result of

egg loss during extraction protocols, detection failure during examination, and due to the fact that direct rectal collections were not possible.

Epidemiological studies predict that parasite diversity, prevalence, and intensity should correlate positively with density within host populations (Anderson and May 1978; Anderson 1979; Mborá and McPeck 2009), as was the case in our investigation. Our results were consistent with several case studies examining host-parasite assemblages within primate populations persisting within forest fragments. These investigations have found that the intensity of habitat disturbance and degree of fragmentation best explain changes in host-parasite dynamics measured by increases in parasite prevalence and species richness (Gillespie and Chapman 2006; Mborá and McPeck 2009)

Small populations with restricted distributions were particularly threatened by increases in pathogenicity or abundance of generalist parasites (Holmes 1996), and in several primate species, population declines within fragmented habitats corresponded directly to these changes in host parasite dynamics (Gillespie and Chapman 2008). Elevated host densities enhance transmission rates by increasing contact rates between individuals as well as between individuals and infective stages in their environments, resulting in positive correlations between density, prevalence, and diversity (Gillespie and Chapman 2006; Puttker, Meyer-Lucht et al. 2008) In both the current investigation as well as several primate systems, parasite diversity and abundance were positively correlated with animal density (Mborá and McPeck 2009). Rubin *et al.* (2002) reported that bighorn sheep utilizing grazing terrain near metropolitan areas had unique internal



parasites not found elsewhere in their range, which is comparable to the isolation of *Capillaria sp.* to elk in urban centers of the Bow Valley (*i.e.* Banff and Canmore).

While this study was not specific towards examining host condition through captures or physical exams, I was observant for any obvious signs of physical distress which may be indicative of deterioration of body condition due to the cumulative effects of parasitic load. Our investigation did not suggest negative impacts of parasites based on visual cues such as coat appearance, body condition/emaciation, or indications of gastrointestinal distress such as diarrhea. Further investigations into host condition would be warranted to examine the possibilities that animals with higher intensity infections are experiencing more discrete physical costs associated with parasitism such as decreases in fecundity or rearing ability, or whether the influence of parasite infection on body condition is compensated for by increased food intake (especially when high quality forage is available within town sites). Under wild conditions it would be exceedingly difficult to track food ingestion, but it is possible that the animals with high parasite loads consume more food than those with lower intensity infections in order to accommodate the energetic demands imposed by their parasites (Puttker, Meyer-Lucht et al. 2008). If this is the case, then current levels of parasitism in the high density herds (*i.e.* Canmore, Banff) may be in equilibrium with the potential for high quality and abundant forage from anthropogenic food sources such as the golf courses or landscaped vegetation. Should the food availability decline or increases in host density raise parasite loads, these populations could potentially display reductions in body condition and concurrent reductions in fitness, survival, and reproduction.

Urbanization and changes in land use are a driving force increasing the degree and rate of interaction between wildlife, domestic animals, and people (Thompson, Kutz et al. 2009). In consideration of the continued recreation and development within the Bow Valley, as well as the potential impact parasitic species can have on their hosts. Further investigations are warranted to continue to monitor the dynamics of host parasite assemblages as well as identifying potential zoonotic species within the Bow Valley population.


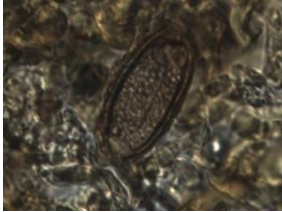
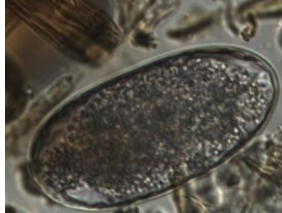

#### 4.5 Tables

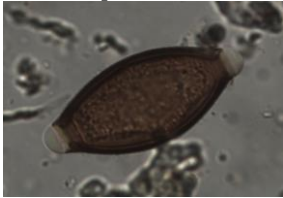
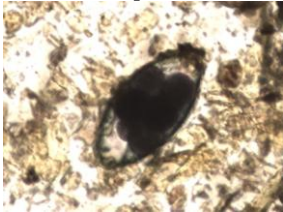
**Table 4-1** Several parasites of potential significance infecting North American elk (*Cervus elaphus*) in Western Canada

<i>Species</i>	<i>Host Tissue</i>	<i>Identification Method</i>	<i>Source</i>
<b><u>Protozoa</u></b>			
<i>Giardia sp.</i>	Duodenum	Not Tested	<b>3</b>
<i>Eimeria sp.</i>	Small Intestine	Wisconsin Fecal Flotation	<b>1</b>
<b><u>Nematoda</u></b>			
<i>Marshallagia marshalli</i>	Abomasum	Wisconsin Fecal Flotation	<b>1</b>
<i>Mazamastrongylus odocoilei</i>	Abomasum	Wisconsin Fecal Flotation	<b>2, 10</b>
<i>Trichostrongylus axei</i>	Abomasum	Wisconsin Fecal Flotation	<b>1, 2, 10</b>
<i>Ostertagia sp.</i>	Abomasum	Wisconsin Fecal Flotation	<b>1, 10</b>
<i>Spiculopteria boehmi</i> <sup>1</sup>	Abomasum	Wisconsin Fecal Flotation	<b>2</b>
<i>Nematodirus helvetianus</i>	Small Intestine	Wisconsin Fecal Flotation	<b>1, 2, 10</b>
<i>Cooperia oncophora</i>	Small Intestine	Wisconsin Fecal Flotation	<b>1, 2, 10</b>
<i>Oesophagostomum venulosum</i>	Large Intestine	Wisconsin Fecal Flotation	<b>2, 10</b>
<i>Dictyocaulus viviparus</i>	Lungs	Flukefinder™	<b>1, 5, 7, 8, 10</b>
<b><u>Trematoda</u></b>			
<i>Fascioloides magna</i>	Liver	Flukefinder™	<b>1, 5, 6, 7, 10</b>
Sources:			
<b>1</b> (Towell and Thomas 2002)	<b>2</b> (deBruyn 2010)	<b>3</b> (Appelbee, Thompson et al. 2005)	
<b>4</b> (Wasel, Samuel et al. 2003)	<b>5</b> (Pybus 1990)	<b>6</b> (Kennedy, Acorn et al. 1999)	
<b>7</b> (Kingscote, Yates et al. 1987)	<b>8</b> (Smits 1991)	<b>9</b> (Corn and Nettles 2001)	

<sup>1</sup>Invasive species endemic to European Red Deer - recently discovered in 3 herds of wild elk in Alberta (deBruyn 2010)

**Table 4-2** Summary of host-parasite assemblages in resident elk throughout the Bow Valley. Pooled data from 2010 and 2011 for Canmore (n=110), BVPP (n=109) and Banff (n=113). Comparisons of prevalence utilized a more stringent significance value of  $p=0.017$  (Dunn- Šidák correction).

Parasite	Location	Prevalence	Prevalence 95% C.I.	Mean Intensity	Intensity 95% C.I.
<b><i>Fascioloides Magna</i></b>  <small>en.wikipedia.org</small>	Banff	83.20%	74.9 to 89.1	23.7	25.1 to 42.0
	Canmore	73.60%	64.6 to 81.2	14.1	11.1 to 18.8
	BVPP	49.50%	40.0 to 59.2	7.8	6.00 to 10.9
<b><i>Capillaria sp.</i></b> 	Banff	52.20%	42.9 to 61.5	12.2	8.9 to 16.5
	Canmore	49.10%	39.5 to 58.7	3.1	2.3 to 4.7
	BVPP	0.00%	0.0 to 4.7	0 <sup>A</sup>	0
<b><i>Trichostrongylus sp. "A"</i></b> 	Banff	65.50%	56.2 to 74.0	5.6	4.6 to 6.8
	Canmore	55.50%	45.9 to 64.6	5.8	4.2 to 8.6
	BVPP	31.20%	22.8 to 40.8	2.1	1.7 to 2.6
<b><i>Trichostrongylus sp. "B"</i></b> 	Banff	31.00%	22.9 to 40.2	2.7	2.1 to 4.0
	Canmore	20.9%	14.0 to 29.5	2.1	1.5 to 3.6
	BVPP	8.30%	4.2 to 15.0	1.7	1.1 to 2.6

Parasite	Location	Prevalence	Prevalence 95% C.I.	Mean Intensity	Intensity 95% C.I.
<i>Trichuris sp.</i> 	Banff	36.30%	27.8 to 45.6	3.2	2.5 to 3.9
	Canmore	16.40%	10.3 to 24.5	1.6	1.3 to 2.0
	BVPP	7.30%	3.5 to 14.1	2.4	1.5 to 3.0
<i>Nematodirus sp.</i> 	Banff	0.90%	-	-	-
	Canmore	3.60%	-	-	-
	BVPP	3.70%	-	-	-

A – No *Capillaria sp.* eggs were recovered from BVPP fecal samples.

B – *Nematodirus sp.* eggs were recovered in trace amounts and were not included in statistical investigations

**Table 4-3** Comparisons of prevalence (Fishers Test,  $p_{crit} = 0.017$ ), and intensity (Bootstrapped t-test,  $p_{crit} = 0.017$ ) within parasite assemblages of elk in the Banff, Canmore, and BVPP herds.

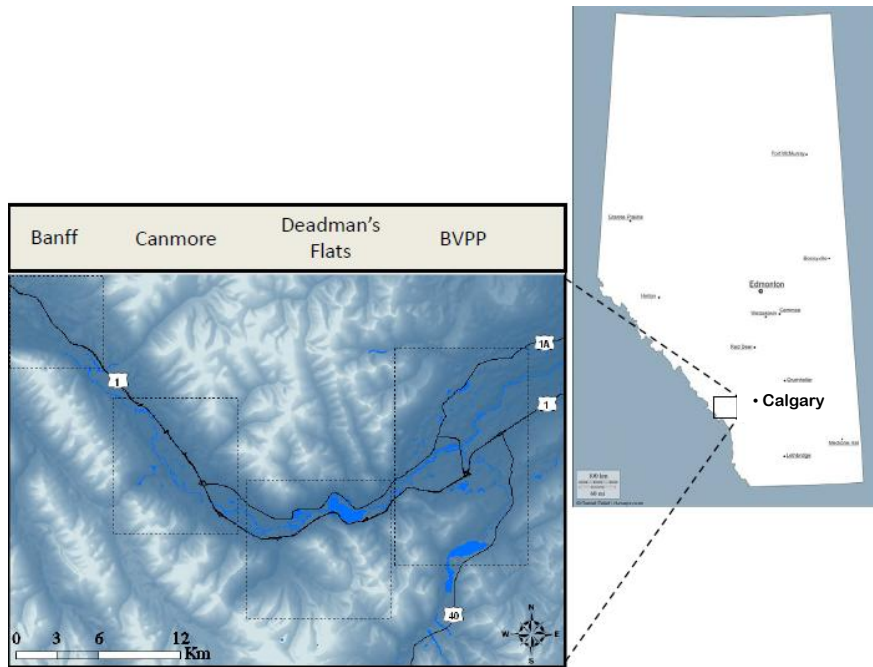
<i>Fascioloides magna</i>	Banff Vs. Canmore	Banff Vs. BVPP	Canmore Vs. BVPP
Prevalence (Fishers Test)	$p = 0.103$	<b><math>p &lt; 0.001</math></b>	<b><math>p &lt; 0.001</math></b>
Intensity (Bootstrapped T-test)	$t = 3.92$ <b><math>p = 0.0005</math></b>	$t = 5.559$ <b><math>p &lt; 0.0001</math></b>	$t = -2.701$ <b><math>p = 0.0110</math></b>
<i>Capillaria sp.</i>	Banff Vs. Canmore	Banff Vs. BVPP	Canmore Vs. BVPP
Prevalence (Fishers Test)	$p = 0.689$	- <sup>A</sup>	-
Intensity (Bootstrapped T-test)	$t = 4.381$ <b><math>p &lt; 0.0001</math></b>	-	-
<i>Trichostrongylus sp. "A"</i>	Banff Vs. Canmore	Banff Vs. BVPP	Canmore Vs. BVPP
Prevalence (Fishers Test)	$p = 0.134$	<b><math>p &lt; 0.001</math></b>	<b><math>p = 0.003</math></b>
Intensity (Bootstrapped T-test)	$t = -0.174$ $p = 0.8480$	$t = 5.732$ <b><math>p &lt; 0.0001</math></b>	$t = -3.492$ <b><math>p = 0.0140</math></b>
<i>Trichostrongylus sp. "B"</i>	Banff Vs. Canmore	Banff Vs. BVPP	Canmore Vs. BVPP
Prevalence (Fishers Test)	$p = 0.095$	<b><math>p &lt; 0.001</math></b>	<b><math>p = 0.012</math></b>
Intensity (Bootstrapped T-test)	$t = 0.941$ $p = 0.3955$	$t = 1.702$ $p = 0.1100$	$t = -0.697$ $p = 0.5235$
<i>Trichuris sp.</i>	Banff Vs. Canmore	Banff Vs. BVPP	Canmore Vs. BVPP
Prevalence (Fishers Test)	<b><math>p = 0.001</math></b>	<b><math>p &lt; 0.001</math></b>	$p = 0.059$
Intensity (Bootstrapped T-test)	$t = 3.765$ <b><math>p = 0.001</math></b>	$t = 1.465$ $p = 0.1390$	$t = 1.642$ $p = 0.1300$

<sup>A</sup> - *Capillaria sp.* was not detected in BVPP samples, and these comparisons were not investigated.

**Table 4-4** AIC scores and ranks for generalized linear models fitting *F.magna* fecal egg counts and parasite assemblage diversity.

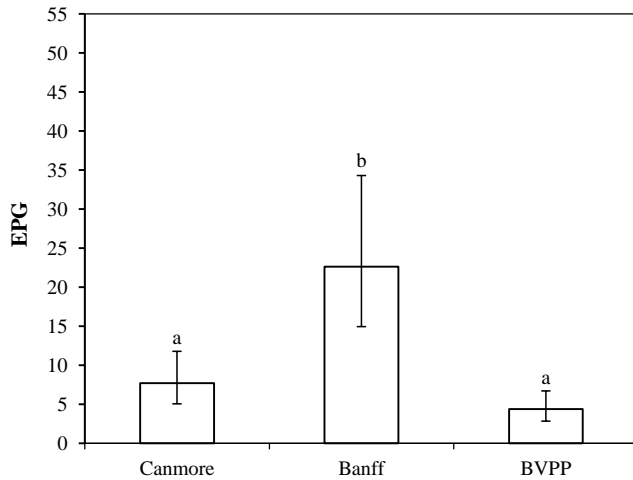
<b>2010</b>				<b>2011</b>			
<i>F.magna</i>	AIC <sub>c</sub>	ΔAIC <sub>c</sub>	Rank	<i>F.magna</i>	AIC <sub>c</sub>	ΔAIC <sub>c</sub>	Rank
locationcode + season + locationcode*season	1014.531	7.3663	3	<b>locationcode + season + locationcode*season</b>	<b>1130.07</b>	<b>0</b>	<b>1</b>
<b>locationcode+season</b>	<b>1008.34</b>	<b>1.1727</b>	<b>1</b>	locationcode+season	1137.698	7.6234	3
<b>locationcode</b>	<b>1007.16</b>	<b>0</b>	<b>1</b>	locationcode	1136.784	6.7099	2
season	1033.566	26.4014	4	season	1163.15	33.0754	4
<b>Diversity</b>	AIC <sub>c</sub>	ΔAIC <sub>c</sub>	Rank	<b>Diversity</b>	AIC <sub>c</sub>	ΔAIC <sub>c</sub>	Rank
<b>locationcode + season + locationcode*season</b>	<b>428.317</b>	<b>0</b>	<b>1</b>	<b>locationcode + season + locationcode*season</b>	<b>514.003</b>	<b>0.2744</b>	<b>1</b>
locationcode+season	431.8247	3.5081	2	<b>locationcode+season</b>	<b>513.728</b>	<b>0</b>	<b>1</b>
locationcode	468.6499	40.3333	3	locationcode	523.9657	10.2374	3
season	502.9568	74.6402	4	season	582.9234	69.1951	4

## 4.6 Figures

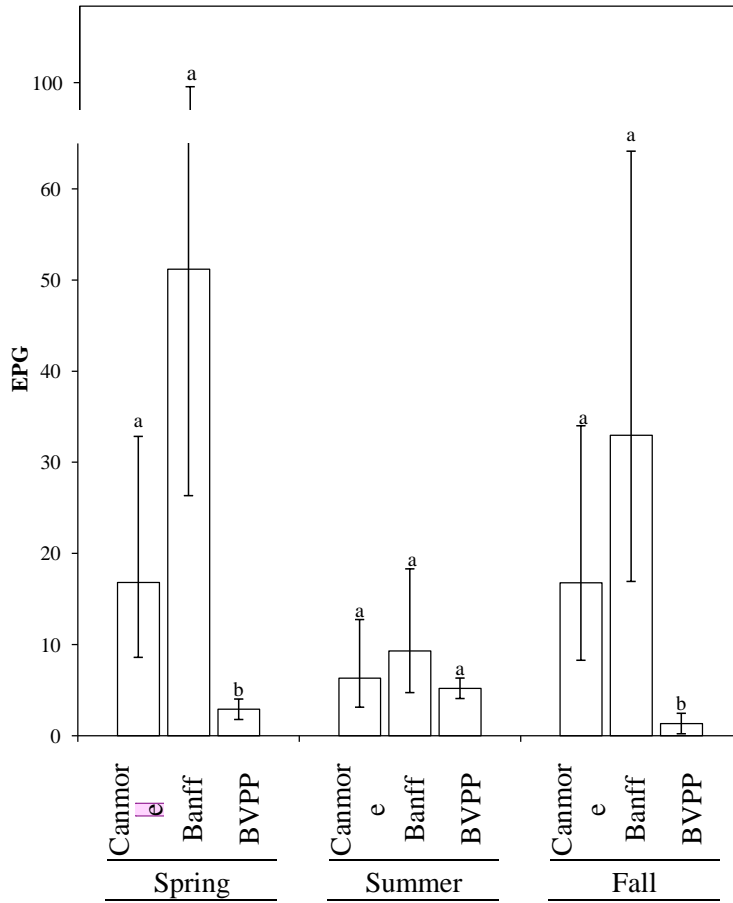


**Figure 4-1** Overview of the Bow Valley corridor study site in Alberta, Canada. Approximations of regional boundaries delineating herd identifications are shown within dashed polygons (Banff, Canmore, Deadman's Flats, BVPP). Alberta overview graphic free sourced courtesy [www.d-maps.com/carte.php?num\\_car=23504&lang=en](http://www.d-maps.com/carte.php?num_car=23504&lang=en)





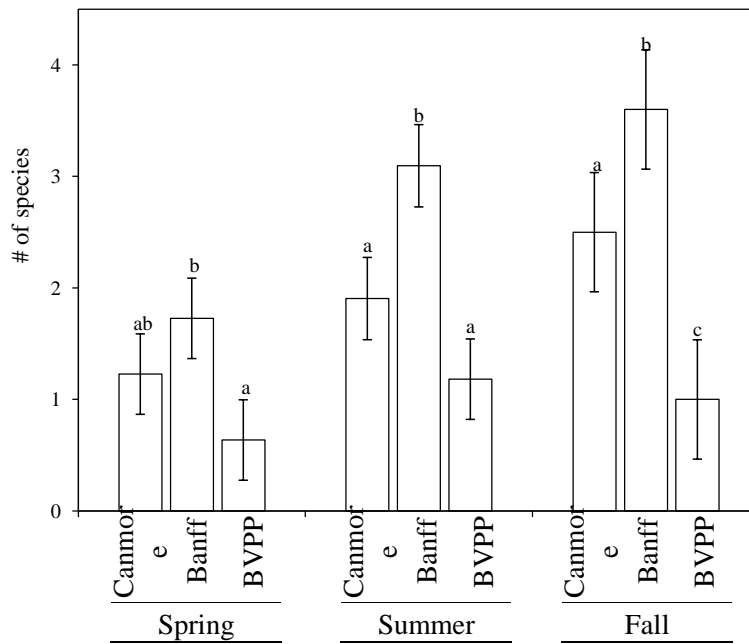
**Figure 4-2** Average (with 95% confidence intervals) *Fascioloides magna* fecal egg counts sampled from the Banff, Canmore, and Bow Valley Provincial Park elk herds during Spring, Summer, and Fall 2010. Backtransformed least-squares means based on top ranked GLM model estimates are presented along with Z-scored test outcomes to indicate significance of pair wise comparisons between locations (only compared within seasons). Dunn-Šidák corrections for multiple comparisons were used to control for inflation of Type 1 errors ( $k=9$ ,  $p^1= 0.0057$ ). Letter above the bar indicates significant differences between sites when the letter is not identical.



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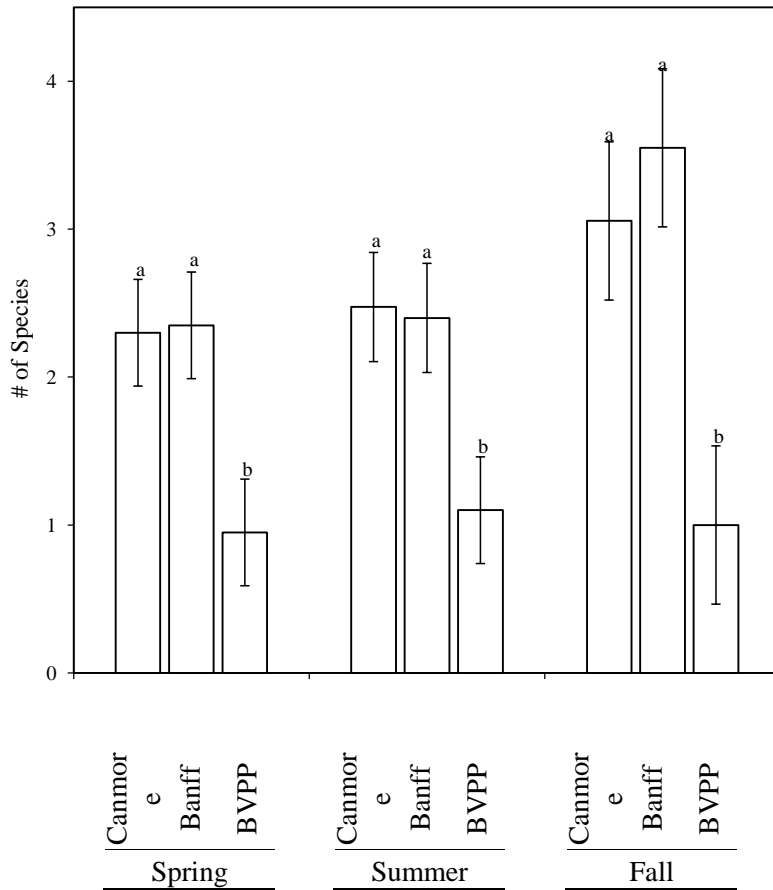
**Figure 4-3** Average (with 95% confidence intervals) *Fascioloides magna* fecal egg counts recovered from the Banff, Canmore, and Bow Valley Provincial Park elk herds during spring, summer, and fall 2011. Backtransformed least-squares means based on top ranked GLM estimates are presented along with significance of pair wise comparisons between locations (only compared within seasons). Dunn-Sidak corrections for multiple comparisons were used to control for inflation of Type 1 errors ( $k=9$ ,  $p^1=0.0057$ ). Letters above the bar indicate significant differences between seasons and sites when the letter is not identical.

**Comment [KR2]:** M sands for model so I am not sure you need to add "model here"



**Figure 4-4** Average parasite diversity (with 95% confidence intervals) recovered from the Banff, Canmore, and Bow Valley Provincial Park elk herds during spring, summer, and fall 2010. Backtransformed least-squares means based on top ranked GLM estimates are presented along with significance of pair wise comparisons between locations (only compared within seasons). Dunn-Šidák corrections for multiple comparisons were used to control for inflation of Type 1 errors ( $k=9$ ,  $p^1=0.0057$ ). Letters above the bar indicate significant differences between seasons and sites when the letter is not identical.

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**Figure 4-5** Average parasite diversity (with 95% confidence intervals) detected within the Banff, Canmore, and Bow Valley Provincial Park elk herds during spring, summer, and Fall 2011. Backtransformed least-squares means based on top ranked GLM estimates are presented along significance of pair wise comparisons between locations (only compared within seasons). Dunn-Šidák corrections for multiple comparisons were used to control for inflation of Type 1 errors ( $k=9$ ,  $p^1= 0.0057$ ). Letters above the bar indicate significant differences between seasons and sites when the letter is not identical.

## 5 Conclusions

### 5.1 Goals of the Study

Overall, this thesis aimed to provide an example of the applied use of home range estimates and resource selection functions in the context of an active land and wildlife management scenario utilizing local elk. Furthermore, this study investigated current host parasite assemblages in the Bow Valley elk as a measure of one of the potential outcomes of the effects of urbanization, habitat fragmentation, and resource use.

In Chapter 3, I utilized high quality and precise GPS derived data to estimate home ranges at the individual and herd-level, and to derive utilization distributions for these animals to describe habitat use at the home range level. Utilization distributions resulting from this analysis were contrasted against locally defined habitat patches and movement corridors to gauge the effectiveness of corridor design and land sequestering in protected foraging patches. GPS data was also utilized to fit a used/available resource selection function (RSF) to spatial distributions of elk in urban (Canmore) and rural (BVPP Deadman's Flats) elk to identify herd level trends in parameters determining habitat selection, as well as differences in resource selection between the urbanized and wild herds.

Chapter 4 considered host-parasite assemblages as a potential indicator of ecological change and focused on creating measures of host parasite assemblages of elk herds within the Bow Valley, for the purpose of establishing baseline surveys to evaluate future changes in host-parasite systems resulting from further ecosystem disruption. I further determined significant differences in infection status between host groups, and

identified parasites of potential clinical or public health significance or clinical concerns. Higher rates and intensities of parasite infections within the townsites were expected largely due to higher densities of infectious parasite stages accumulating in the environment due to repeated and sustained use of a small number of attractive habitat/foraging patches (*i.e.* golf courses and other anthropogenically modified terrain suitable for grazing), as well as smaller overall home ranges limiting habitat choice and condensing animals.

The utility of considering parasites as an indicator of ecological change is not limited to elk, but any group of animals choosing to exploit resources in proximity to urban centers or changing environments including, but not limited to: coyote (Catalano S., Lejeune M. et al. 2012), bighorn sheep (Rubin, Boyce et al. 2002), musk ox (*Ovibos moschatus*) (Kutz, Hoberg et al. 2004), caribou (Kutz, Hoberg et al. 2004) or feral/domestic canids and felines (Joffe, Van Niekerk et al. 2011).

### **Summary of Major Findings**

Overall, home range of the Bow Valley elk was greatly constrained by the mountainous topography along the corridor. Home range size was small across all herds, but total size was smaller and core use areas more focused in urban animals in Canmore and highly fragmented and constrained animals utilizing the Deadman's Flats area. Bow Valley elk were less constrained by steep mountain slopes and urban development and had greater total home range size and more dispersed core area. Migratory behavior was not observed in these animals and home range extent did not change between the seasons, while resource selection functions identified the same model for both winter and summer,

although model parameters differed between urban (Canmore) and rural (Deadman's Flats/Bow Valley Provincial Park) animals. The home range models identified a lack of overlap between the established corridor/habitat patch boundaries and animal home range polygons. This suggests the corridor lacks functionality for species such as elk, and home range and resource selection models could provide better baseline datasets for the establishment and protection of additional wildlife corridors.

In proportion to habitat availability, urban animals selected against steep slopes, high elevations, cover habitat (white spruce), and high road density, while selecting for grazing habitat on town margins. Rural animals showed similar selection against steep slopes and high road densities and in favor of grazing habitat types, but also selected for cover habitat (shrubland, white spruce & trembling aspen).

Fecal surveys identified a number of parasites in the Bow Valley elk, including: liver flukes (*Fascioloides magna*), protozoan (*Eimeria sp.*), and gastrointestinal nematodes (*Capillaria sp.*, *Nematodirus sp.*, *Trichostrongylus sp.*, *Trichuris sp.*). Overall, parasite prevalence, intensity, and diversity were greatest in the urban centers (Banff and Canmore) and smallest in rural samples (Bow Valley Provincial Park).

## **5.2 Animal Mortality & Prevention Strategies**

Animal mortality within the Town of Canmore was low and none of the collared animals were lost to natural or human related causes over the duration of the study. Among the eleven collared animals within the BVPP and Deadman's Flats herds, three were killed on the Trans-Canada Highway (Hwy. 1) as a result of vehicle collisions, and another three were lost to predation by cougars or wolves. The high contrast in

survivorship compared to the Canmore animals is likely due to the lack of preventative wildlife fencing along much of the Trans-Canada Highway from BVPP to DMF, as well as the effectiveness of the town site as a refuge from predation. The high highway mortality of the BVPP animals is due to the attractive grazing habitat along highway right of ways and medians, the lack of wildlife control measures (*i.e.* fencing) and high rates of travel along the Trans-Canada highway, which bisects Northern and Southern portions of the BVPP home range. Perhaps the most important discovery of this study is the need for (1) preventative wildlife fencing and appropriate highway crossing structures (Clevenger and Waltho 2005) within the eastern edge of the Bow Valley corridor; and (2) greater emphasis on speed control through legislation or enforcement within identifiable “high-risk” areas in Bow Valley Provincial Park.

Other incidents to note included a mass mortality event in BVPP in which a small herd of ~14 cow elk moved onto thin ice on Middle Lake and fell through the ice. This event resulted in mortalities of 7 mature cows of which two had gestating calves. This event is reminiscent of the extirpation of the remaining Banff National Park woodland caribou (*Rangifer tarandis*) in a single natural die-off due to a single avalanche in 2009 (Hebblewhite, White et al. 2009). The effects of a natural disaster such as this; when compounded by existing stress from anthropogenic sources or habitat fragmentation, can have devastating effects on a struggling population.

Finally, in early 2013, two mature bulls of dominant reproductive status within the Town of Canmore were euthanized by Alberta Parks (ATPR) biologists, due to critical trauma likely sustained during the preceding mating season. Perforating cranial injuries and evidence of thoracic hematoma/rib fractures suggests that these males were



likely injured while sparring (ATPR – Personal Communication). The simultaneous loss of these breeding males presents an interesting situation in terms of genetic fitness of future generations sired by formerly subordinate males which will emerge in future rutting seasons due to the greater availability of mating opportunities.

### **5.3 Future Directions**

Concentration of elk on reduced habitat patches has the potential to contribute to major detrimental effects on local vegetation crops; for example, in cases where intense herbivory by elk has been found to reduce standing biomass of willows (Baker, Ducharme et al. 2005) and aspen (White, Feller et al. 2003) The loss of trophic cascades originating from wolf predation on elk has been suggested as a leading driver of Aspen depletion within Yellowstone National Park and elsewhere in the Rocky Mountains (Fortin, Beyer et al. 2005). This suggests that; if left unchecked by natural predation, the urban elk within the Town of Canmore have the potential to significantly affect the local vegetation (both forage and browse species) through their foraging activity. The high-density herd within the Canmore area could be an ideal system to examine grazer-vegetation trophic cascades, which are to a certain extent exempt from regulation through the impact of large carnivores.

On a broader scale, the identification and integration of protected local and regional wildlife corridors is a fundamental component of wildlife management and conservation by allowing connectivity of habitat between residual habitats in fragmented environments (Beier, Penrod et al. 2006; Chetkiewicz and Boyce 2009). A typical approach to corridor identification and management is Least Cost Path (LCP) analysis

which evaluates potential routes across landscapes based on the “cost” of animal movement estimated through the influence of physical and natural features such as slope, elevation, linear obstructions, anthropogenic influences, cover potential, and others (Theobald 2006). This type of cost modeling requires that the local landscape be defined as a gradient along a scale of selection, the parameterization of which is usually achieved through expert opinions (Chetkiewicz and Boyce 2009). These can potentially lead to misinterpret species-specific recommendations for corridor parameters and provide ineffective or non-functional corridor estimates. Such is the case in the 1991 Golder Associates report for Three Sisters Mountain Village, advocating the use of non-winter forestry buffer zones, in which (annual) multi-species corridor estimates were based on data for deer and elk in remote (*i.e.* low human use) regions of Washington and Oregon (Herrero 2000) These estimates are inadequate to promote habitat connectivity and animal migration along the Bow Valley in the vicinity of Canmore (Herrero 2000).

Chetkiewicz and Boyce (2009) reported a methodology for Least Cost Path (LCP) analysis to identify potential conservation corridors for carnivores in the Bow Valley and Waterton conservation areas, which utilized RSF estimates to construct cost surfaces for corridor placement. By utilizing RSF values when delineating corridor boundaries, their approach allows the integration of the functional connectivity of landscapes based on robust estimates of animal resource selection, with determinations of corridor effectiveness, which reflect seasonal and species-specific variances in resource selection (Boyce 2006; Chetkiewicz and Boyce 2009). These empirical estimates provide species and landscape specific gradients of selection, which replace the reliance on expert opinion or generalized standards of corridor construction with data-based and location-

specific approaches to parameterize LCP cost surfaces (Manly, McDonald et al. 2002; Noss and Daly 2006; Beier, Majka et al. 2008; Chetkiewicz and Boyce 2009). This approach allows for a more “rigorous, defensible and transparent” corridor design (Chetkiewicz and Boyce 2009). As a well documented and supported method for the construction of conservation corridors for carnivores in the Bow Valley, LCP analysis as per Chetkiewicz and Boyce (2009) would provide the next logical step in promoting effective management strategies and ensuring viable habitat continuity for elk and other large ungulate species throughout the Bow Valley corridor.

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## **Appendix A: Software & Open-Source ArcGIS Tools**

### **Commercial Software**

ESRI ArcGIS for Desktop Advanced - Version 10.1 (ESRI 2012 Redlands, CA)

<http://resources.arcgis.com/en/home/>

ESRI ArcGIS Version 9.3 - ArcInfo License (ESRI 2012 Redlands, CA)

<http://resources.arcgis.com/content/arcgisdesktop/9.3/about>

SAS® 9.3 (SAS Institute Inc. 2012 Cary, NC)

<http://www.sas.com/software/sas9/>

### **Open-Source Programs and Extensions for ArcGIS Software Suites**

Hawth's Spatial Analysis Tools Version 3.27 ( 2007 Hawthorne Beyer)

An extension for ESRI's ArcGIS (ArcMap) software suite, providing tools for spatial manipulations and functions common within the context of ecological applications such as animal movement analysis. This extension has been officially discontinued and replaced by the open-source Geospatial Modeling Environment software and will not function with ESRI products more recent than 9.3. Compatibility issues have arisen in recent versions of the 9.x build of the ESRI ArcMap program.

<http://www.spatial ecology.com/htools/download.php>

GME (2012 Hawthorne Beyer, Spatial Ecology LLC)

The Geospatial Modeling Environment (GME) is the suite of spatial analysis and modeling tools which replaces its predecessor: Hawth's Tools. This open-source framework provides a toolbox to utilize the R statistical engine to implement spatial analyses under typical ecological approaches. This toolbox requires R (2.15.0), StatConn DCOM, and ESRI ArcGIS (ArcView license or higher for 10.1) but can be run external to the ESRI software suite.

<http://www.spatial ecology.com/gme/gmedownload.htm>

R version 2.15.0 (2012 The R Foundation for Statistical Computing)

Open source statistical computing package.

<http://www.r-project.org/>

LoCoH for ArcGIS/ArcView (Wayne Getz, 2005, Berkley CA)

Local convex hull home range estimate generator. Runs as an ArcMap toolbox using the R computational engine.

<http://locoh.cnr.berkeley.edu/arctutorial>

Quantitative Parasitology 3.0 (2005 Reiczigel, J.; Rozsa, L. Budapest, HUN)

This parasitology software provides statistically correct ways to analyze the highly aggregated (right-skewed) frequency distributions exhibited by parasites. QP 3.0 is recommended to describe parasitic infections within a

sample of hosts and to compare parasitic infections across different  
samples of hosts. [www.zoologia.hu/qp](http://www.zoologia.hu/qp)

## **Appendix B: Laboratory Protocols (Parasite Egg Extraction)**

### **1) Wisconsin Double Centrifuge Technique**

*Source: WCVM Parasitology Diagnostic Techniques Handbook, March 2002 Courtesy Dr. S. Kutz, University of Calgary (Faculty of Veterinary Medicine)*

#### **Introduction**

This technique is used to concentrate and count parasite eggs and oocysts in feces. It is based on the principle that the eggs and oocysts are less dense than a sugar flotation solution with a high specific gravity and will therefore rise to the top of a feces/flotation solution mixture where they can be collected, examined and counted microscopically.

#### **Materials**

Wax coated paper cups

Wooden tongue depressors

Cheesecloth (40-60wt) cut into 20x20 cm squares

16x125mm glass test tubes

Microscope slides

Glass coverslips (22x22mm)

Centrifuge with swing arm rotor (IEC HN-S)

Compound Microscope with 10x and 40x objectives

Scale or balance capable of weighing 4g fecal samples

Permanent marker

Plastic wash bottle

Dropper bottle

Syringe or graduated cylinder capable of measuring and dispensing 12ml



## Reagents

Tap water

Sheather's flotation solution (specific gravity 1.26)

### Sheather's Recipe

39.66 ml Formaldahyde

3000g white sugar

2436 ml distilled water

*Gives 4000ml Sheather's flotation solution*

## Procedure

- 1) For each fecal sample stack two paper cups and label the outside cup, using a permanent marker, with the laboratory identification number (the inside cup is A the outside cup is B)
- 2) Use a tongue depressor to weigh 4g of feces sample into cup A
- 3) Add approximately 12ml of tap water to each sample (if feces are very dry allow to soak 5-10 minutes)
- 4) Use a tongue depressor to mix the feces and water until homogeneous
- 5) Pull a single layer of cheesecloth over the top of cup B. Squeeze cup a to make a spout and pour the fecal mixture through the cheesecloth into cup B.
- 6) With a wash bottle, rinse cup A with approximately 3ml of tap water and pour through cheesecloth filter into cup B.
- 7) Pour the filtrate into a labeled 16x125mm test tube.
- 8) Centrifuge the tube for 5 minutes at 1500rpm
- 9) After Centrifugation, using a single pouring motion, decant the supernatant taking care not to disturb the sediment

- 10) Add 4-5ml of Sheather's solution to the sediment, mix well using a vortex mixer and then fill the test tube with Sheather's to 5mm from the top.
- 11) With a dropper bottle add Sheather's solution until there is a slightly convex meniscus
- 12) Carefully place a 22x22mm coverslip on the meniscus in the tube.  
*\*\*Technical note: avoid trapping large air bubble under the coverslip. If such bubbles form attempt to break them using the corner of the coverslide before placing it on the meniscus.*
- 13) Centrifuge the tube for 5 minutes at 1500 rpm.  
*\*\*Technical note: If after centrifugation, bubbles occupy more than 30% of the area under the coverslip repeat the procedure with remaining sample. If this is not possible, note the extent of bubbles under the coverslip and record in the comments section of the data sheet.*
- 14) Lift the coverslip straight up and transfer to an appropriately labeled slide. Do not slide the coverslip along the top of the tube
- 15) Examine the slide under 100x total magnification. Count and differentiate all parasite stages (eggs and oocysts) and record results on, and initial, the data sheet. Results are recorded as eggs per 4g of feces. Use the comments section of the data sheet to report concerns such as: the presence/absence of protostrongylid larvae, the size of pellets (e.g. small calf); and any sample quality concerns, such as bubbles, old or dry feces, larvated nematode eggs, sporulated cysts, and arthropod eggs.

## 2) **Fluke-Finder**

### **Introduction**

The Fluke-Finder (FlukeFinder™ Moscow, ID) apparatus consists of a sequential two-stage columnar sieve designed to mechanically separate detritus (coarse/fibrous plant material) from larger parasite eggs (i.e. liver flukes and other trematoda) contained in fecal collections. Protocols were provided by the lab of Dr. S. Kutz, Department of Veterinary Medicine, University of Calgary.

### **Materials**

Wax coated paper cups

Wooden tongue depressors

FlukeFinder™ apparatus

Tap water

100mm plastic petri dish (Draw 10mm x 10mm search grid on underside of dish)

Dissecting microscope

Permanent marker

Plastic wash bottle

### **Procedure**

- 1) For each fecal sample, label one cup, using a permanent marker, with the laboratory identification number
- 2) Use a tongue depressor to weigh 2g of feces sample into cup
- 3) Add approximately 12ml of tap water to each sample (if feces are very dry allow to soak 5-10 minutes)
- 4) Use a tongue depressor to mix the feces and water until homogeneous

- 5) Wash homogenous mixture into the top collecting section of the Fluke-finder under a low intensity stream of cool tap water
- 6) Continue to rinse sample for 90 seconds under a low intensity stream of cool tap water. Eggs will pass through primary sieve and collect above the secondary sieve. High intensity/volume water flow will result in water buildup between sieves and loss of filtrate through breather ports between sieves.
- 7) The second (lower) collection screen was rinsed into a collection beaker using the minimum volume of water necessary to adequately collect filtrate
- 8) The volume of water + filtrate was transferred to a 100mm diameter plastic petri dish (multiple dishes if necessary) and examined using a dissecting microscope (30x magnification).
- 9) Search grids drawn on underside of petri dishes (10mm x 10mm grid) facilitate visual sweeps.
- 10) Eggs within the volume were counted and recorded. Final fecal egg counts (FEC) are reported as eggs per gram (EPG).