Disclaimer

This report presents preliminary findings from the 2008 research program within the Foothills Research Institute (FRI) Grizzly Bear Program. It must be stressed that these data are preliminary in nature and all findings must be interpreted with caution. Opinions presented are those of the authors and collaborating scientists and are subject to revision based on the ongoing findings over the course of these studies.

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INTRODUCTION

Gordon Stenhouse¹

¹ Foothills Research Institute Grizzly Bear Program (FRIGBP)

Introduction

Over the past ten years the Foothills Research Institute’s Grizzly Bear Program has made significant advances in improving our understanding of how grizzly bears use forested landscapes within their range in Alberta. Some of this information has been recently used by Alberta Sustainable Resource Development (SRD) to delineate new grizzly bear management zones (core and secondary habitats) along the eastern slopes. We have recently published a new research paper which details the methodologies used to determine what areas would meet the specified criteria to act as these new conservation areas.

Our research program continues with the development of new tools and models to assist in sustainable forest and land management practices and decisions concerning the long-term conservation of grizzly bears.

A multi-disciplinary team of researchers from the Foothills Research Institute, University of Alberta, University of Calgary, University of Saskatchewan, and Wilfred Laurier University have developed these tools and models. With the support of our program partners (See Appendix 1 & 2) we have now updated all these products so that they are based on specific ecosystem units and recognized bear management areas within Alberta. This year we have completed a new training course to allow all our research partners and interested stakeholders to learn not only how to conduct analysis with these tools, but also how to interpret the results of these analyses. We plan to hold at least three training sessions in 2009.

In 2008, we embarked on the development of new food based RSF models for the Chinchaga/Clear Hills area. A first draft of this new product will be provided in the spring of 2009. Preliminary work to determine the use of whitebark pine by grizzly bears in selected higher elevation sites was initiated in 2008. Projects continuing from last year include examining mountain pine beetle forestry activities and its impacts on grizzly bears and the relationship between weather on grizzly bear’s denning activities and habitat use. Our remote sensing teams have been busy with ongoing research and production of landscape/habitat maps and tracking landscape change. We have also made significant progress in our ongoing work on documenting and defining grizzly bear health conditions along the eastern slopes. These health condition data have been improved by the inclusion of new laboratory results from the pioneering work we have undertaken on the measurement of chronic stress in grizzly bears. Interest in this work has been received from researchers in other areas of North America and from Scandinavian researchers.
These scientists are now interested in similar measurements for the species they are studying which requires our team to figure out new ways to share our laboratory techniques.

The results of our research efforts will enable all our program partners to make well-informed and timely land use and planning decisions to maintain ecosystem health and to support sustainable development in provincial grizzly bear habitat. Although this project focuses on Alberta grizzly bear populations, the concepts, techniques and relationships uncovered can be applied to a variety of species at risk in Alberta and Canada. Our new techniques and achievements are now being tested on polar bears in the Canadian arctic, grizzly bears in Yellowstone and caribou in Alaska and British Columbia. These leading edge innovative products and techniques will make Alberta a recognized world leader in ecosystem management and monitoring.

This annual program report for the Foothills Research Institute’s Grizzly Bear Program (formerly known as the Foothills Model Forest Grizzly Bear Research Program) is divided into separate sections which provide detail on the various program elements within the research effort. These sections have been prepared by the principal investigators of these elements who have or will be publishing most results in scientific peer reviewed journals. A listing of research publications is presented in Appendix 3. Our research team had a great year in 2008, writing a large number of peer reviewed publications along with many that have been submitted.

On behalf of our research team we want to extend our gratitude to every program partner for your ongoing support and encouragement of our work.

Gordon Stenhouse
Karen Graham
CHAPTER 1: PROGRAM CAPTURE ACTIVITIES 2008

Gordon Stenhouse\textsuperscript{1,2} and Bernie Goski\textsuperscript{1,2}

\textsuperscript{1}Foothills Research Institute, \textsuperscript{2}Alberta Sustainable Resource Development, Fish and Wildlife Division

Introduction

The 2008 grizzly bear capture session was the 10\textsuperscript{th} conducted by the Foothills Research Institute (FRI) Grizzly Bear Program previously known as the Foothills Model Forest Grizzly Bear Research Program. In 1999, our original study area encompassed 10,000 km\textsuperscript{2} in an area south of Highway 16, between Edson and Jasper in the north and the Brazeau River in the south. In 2003 the study area expanded to include all of the grizzly bear range between the Berland River and the Montana border (62\% of grizzly bear range in Alberta). In 2005, the study area expanded again to include areas between the Berland and Wapiti Rivers plus the Swan Hills. In 2006, the study area expanded northwards to include the Chinchaga River, the Hotchkiss River and the Meikle River area. We also included an area south of the Wapiti River and returned to the Swan Hills. In 2007 our capture and collaring efforts were focused in the Kakwa and Nose Hill Tower areas of the Weyerhaeuser Forest Management Area (FMA) and the Clear Hills area north of Worsley which included the southern part of the Chinchaga area.

In 2008 our capture and collaring efforts was again focused in the Kakwa River, Nose Hill Tower and Two Lakes areas which is a landbase that represents current MPB outbreak and is part of the Weyerhaeuser FMA. We also captured grizzly bears around the Cheviot mine site southwest of Hinton to continue with ongoing trials and testing of our “animal pathfinder” system, and to collect additional field data for a PhD program at the University of Alberta. One capture also occurred in southwestern Alberta for a MSc. student at the University of Alberta.

The focus of the Kakwa capture effort was to collect grizzly bear movement data that will relate to ongoing mountain pine beetle activities and forest management efforts in this area. Our research team has recently completed an RSF map product for the Kakwa area with grizzly bear location data obtained prior to mountain pine beetle activity. In addition data was needed from collared bears on habitat use and denning selection.

The goal of this year’s capture session was to maintain a sample of approximately ten GPS radio-collars on adult grizzly bears in this area. We also embarked on the second year of deployment of new camera/sensor systems that were attached to GPS collars. These new camera/sensor systems are being used to gather additional information on habitat use and detailed movement paths.
Study Area
The study area where the Kakwa collaring efforts took place is shown in Figure 1.

Methodology
Field capture efforts began in May this year with field crews working out of the Two Lakes and Kakwa Tower areas. The crews consisted of biologists with experience in grizzly bear capture and a project veterinarian. The capture team used a combination of ground based trapping along existing forest access roads to establish culvert and snaring sites as well as some helicopter snare line work in remote areas where access was limited. Trap alarms were used on all snares and culvert traps to minimize the amount of time a bear was held after capture. Primary capture efforts were concluded at the end of June. An aerial capture program to replace collars was conducted in August and October.

We anaesthetized grizzly bears using a combination of xylazine and Telazol administered by remote drug delivery, e.g., dart rifle. Once immobilized, grizzly bears were weighed, and measured (chest girth, zoological length, and straight-line length). Samples were collected (blood, hair, skin biopsy, and tooth). Radio-collar and ear tag transmitters were attached. A transponder was also inserted for future identification purposes. Vital functions and blood-oxygen levels were monitored throughout the handling period. Following handling, we administered atipamezole to reverse the effects of anaesthesia and monitored the grizzly bears until they showed imminent signs of recovery. We re-
checked all bears again within 24 hours of capture to ensure they had recovered fully from immobilization. All details of capture operations conformed to national and provincial standards on the capture and handling of ursids.

Summary of Captures
In total, we captured 21 grizzly bears (Table 1). Ten were caught in the Kakwa/Two Lakes area, one bear was captured in the Clear Hills area as a result of management actions, four bears were captured in the mine areas south of Hinton (one of these bears was a subadult (****) that was released without processing), and two bears were captured in SW Alberta. We also assisted Fish & Wildlife officers in processing three grizzly bears from Hinton, Grande Prairie and Grand Cache and a Banff bear. No black bears or other non-target species were captured this field season. No capture related mortalities occurred during the 2008 field season. Biopsy dart samples of tissue were also collected from seven other grizzly bears which did not involve capture and handling. No cubs of the year were caught. Table 2 lists the female grizzly bears accompanied with cubs.

<table>
<thead>
<tr>
<th>Grizzly Bear IDs</th>
<th>Area</th>
<th>Age</th>
<th>Sex</th>
</tr>
</thead>
<tbody>
<tr>
<td>G017</td>
<td>Coal Valley Mine</td>
<td>Adult</td>
<td>Male</td>
</tr>
<tr>
<td>G077</td>
<td>Waterton N.P.</td>
<td>Adult</td>
<td>Female</td>
</tr>
<tr>
<td>G083</td>
<td>Pincher Creek</td>
<td>Adult</td>
<td>Male</td>
</tr>
<tr>
<td>G102K</td>
<td>Banff N.P.</td>
<td>Adult</td>
<td>Male</td>
</tr>
<tr>
<td>G110</td>
<td>Whitehorse Creek</td>
<td>Sub-Adult</td>
<td>Male</td>
</tr>
<tr>
<td>G111</td>
<td>Whitehorse Creek</td>
<td>Adult</td>
<td>Female</td>
</tr>
<tr>
<td>G223</td>
<td>Kakwa</td>
<td>Adult</td>
<td>Female</td>
</tr>
<tr>
<td>G230</td>
<td>Two Lakes</td>
<td>Adult</td>
<td>Male</td>
</tr>
<tr>
<td>G243</td>
<td>Pincher Creek</td>
<td>Sub-Adult</td>
<td>Male</td>
</tr>
<tr>
<td>G251</td>
<td>Two Lakes</td>
<td>Sub-Adult</td>
<td>Female</td>
</tr>
<tr>
<td>G252</td>
<td>Two Lakes</td>
<td>Sub-Adult</td>
<td>Male</td>
</tr>
<tr>
<td>G253</td>
<td>Kakwa</td>
<td>Adult</td>
<td>Female</td>
</tr>
<tr>
<td>G254</td>
<td>Two Lakes</td>
<td>Sub-Adult</td>
<td>Female</td>
</tr>
<tr>
<td>G255</td>
<td>Grande Prairie</td>
<td>Adult</td>
<td>Female</td>
</tr>
<tr>
<td>G256</td>
<td>Two Lakes</td>
<td>Sub-Adult</td>
<td>Male</td>
</tr>
<tr>
<td>G260</td>
<td>Kakwa</td>
<td>Adult</td>
<td>Female</td>
</tr>
<tr>
<td>G262</td>
<td>Kakwa</td>
<td>Adult</td>
<td>Male</td>
</tr>
<tr>
<td>G264</td>
<td>Kakwa</td>
<td>Adult</td>
<td>Male</td>
</tr>
<tr>
<td>AB-5237</td>
<td>Hinton</td>
<td>Adult</td>
<td>Male</td>
</tr>
<tr>
<td>AB-5187</td>
<td>Grand Cache</td>
<td>Adult</td>
<td>Female</td>
</tr>
</tbody>
</table>
| ****            | Coal Valley Mine   | Sub-Adult | ?????
Table 2: Females and cubs.

<table>
<thead>
<tr>
<th>Grizzly Bear ID</th>
<th>Cubs*</th>
<th>Date</th>
</tr>
</thead>
<tbody>
<tr>
<td>G077</td>
<td>2 COY</td>
<td>May 2/08</td>
</tr>
<tr>
<td>G255</td>
<td>2 yearlings</td>
<td>May 22/08</td>
</tr>
<tr>
<td>G223</td>
<td>2 COY</td>
<td>Aug 29/08</td>
</tr>
<tr>
<td>G260</td>
<td>2 COY</td>
<td>May 2008 - Both COY Deceased in summer 2008</td>
</tr>
</tbody>
</table>

*coy = cubs of the year

**Capture Type**

Capture types were categorized as ground capture with snare, ground capture with culvert trap or heli-darted. Most ground capture sites used snares in three different formats, pail sets, cubby sets and/or trail sets.

Of 21 capture events (Table 3), helicopter captures accounted for 43% (9) of capture events and ground captures accounted for 57% (12) of capture events. Four ground-capture events involved snares, seven involved a culvert trap and one was free range darted from a truck with aerial support.

Table 3. Grizzly bear capture types.

<table>
<thead>
<tr>
<th>Grizzly Bear IDs</th>
<th>Capture Types</th>
<th>Grizzly Bear IDs</th>
<th>Capture Types</th>
</tr>
</thead>
<tbody>
<tr>
<td>G083</td>
<td>Aerial</td>
<td>G110</td>
<td>Aerial</td>
</tr>
<tr>
<td>G243</td>
<td>Culvert</td>
<td>G252</td>
<td>Snare</td>
</tr>
<tr>
<td>G077</td>
<td>Aerial</td>
<td>G253</td>
<td>Free Range Darted from truck</td>
</tr>
<tr>
<td>G251</td>
<td>Snare</td>
<td>G254</td>
<td>Snare</td>
</tr>
<tr>
<td>G255</td>
<td>Culvert</td>
<td>G223</td>
<td>Aerial</td>
</tr>
<tr>
<td>G256</td>
<td>Snare</td>
<td>G260</td>
<td>Aerial</td>
</tr>
<tr>
<td>G262</td>
<td>Aerial</td>
<td>G017</td>
<td>Culvert</td>
</tr>
<tr>
<td>G230</td>
<td>Culvert</td>
<td>AB-5237</td>
<td>Culvert</td>
</tr>
<tr>
<td>G264</td>
<td>Aerial</td>
<td>G102K</td>
<td>Aerial</td>
</tr>
<tr>
<td>G111</td>
<td>Aerial</td>
<td>AB-5187</td>
<td>Culvert</td>
</tr>
<tr>
<td>****</td>
<td>Culvert</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

**Telemetry**

We deployed a GPS radio-collar and ear-tag transmitter on all captured bears except for G083 and G102K. The radio-collars were programmed to collect locations at a rate of 1 per hr. All radio-collars are outfitted with a remote release mechanism in addition to a rot-off system as a backup in case of electronic failure. Radio-collars deployed consisted of 3 types, Tellus (including new UHF model), ATS GPS and Telonics Argos GPS.

We have been doing monthly data upload flights from all collared bears in the Kakwa area. Data collection from other collared bears were being undertaken by the researchers involved. We will continue to collect grizzly bear location and movement data from collared bears up to the time of denning, which usually occurs in November, but is somewhat weather dependent. We are also determining exact locations of all den sites in order to investigate denning behaviour. We will place weather monitoring stations at a
number of grizzly bear dens in December to collect microsite weather data to relate to temperature data being collected from collars in dens (See Chapter 2).

Table 4 gives the status of grizzly bears as of October 2008.

**Table 4. Status of 2008 research grizzly bears as of October 2008.**

<table>
<thead>
<tr>
<th>Grizzly Bear IDs</th>
<th>Fate as of October 2008</th>
</tr>
</thead>
<tbody>
<tr>
<td>G017</td>
<td>Collar was pulled off after 2 days.</td>
</tr>
<tr>
<td>G077</td>
<td>Alive, collar functioning</td>
</tr>
<tr>
<td>G102K</td>
<td>Dead (Shot), collar retrieved</td>
</tr>
<tr>
<td>G110</td>
<td>Alive, collar functioning</td>
</tr>
<tr>
<td>G111</td>
<td>Alive, collar functioning</td>
</tr>
<tr>
<td>G223</td>
<td>Alive, collar functioning</td>
</tr>
<tr>
<td>G224</td>
<td>Alive, collar functioning</td>
</tr>
<tr>
<td>G230</td>
<td>Alive, Collar dropped and recovered</td>
</tr>
<tr>
<td>G238</td>
<td>Collar recovered, Rotted off</td>
</tr>
<tr>
<td>G243</td>
<td>Alive, collar was triggered off in fall</td>
</tr>
<tr>
<td>G251</td>
<td>Alive, collar functioning</td>
</tr>
<tr>
<td>G252</td>
<td>Alive, collar functioning</td>
</tr>
<tr>
<td>G253</td>
<td>Dead (Unknown cause), Collar retrieved</td>
</tr>
<tr>
<td>G255</td>
<td>F&amp;W bear</td>
</tr>
<tr>
<td>G254</td>
<td>Alive, Collar functioning</td>
</tr>
<tr>
<td>G256</td>
<td>Bear not located in fall</td>
</tr>
<tr>
<td>G260</td>
<td>Alive, collar functioning</td>
</tr>
<tr>
<td>G262</td>
<td>Alive, collar malfunctioning.</td>
</tr>
<tr>
<td>G264</td>
<td>Alive, collar malfunctioning.</td>
</tr>
<tr>
<td>G265</td>
<td>Alive, collar on low battery as anticipated</td>
</tr>
<tr>
<td>G266</td>
<td>Alive, collar on low battery as anticipate</td>
</tr>
</tbody>
</table>

**Capture Related Mortalities**

There were no capture related mortalities of either grizzly or black bears this year.

**Black Bears**

No black bears were captured this year.
CHAPTER 2: DENNING, HABITAT SELECTION AND WEATHER 2008

Karine Pigeon\textsuperscript{1,3}

\textsuperscript{1}Foothills Research Institute; \textsuperscript{3}Laval University

\textbf{Introduction}

\textbf{Hibernation}

During 2007-2008 and 2008-2009 winters, we installed HOBO\textsuperscript{®} Micro Station Data Loggers fitted with temperature, relative humidity, solar radiation, wind, and barometric pressure sensors within 500 meters of all accessible collared-grizzly bear dens of the study area (winter 2007-2008 \(n = 6\); winter 2008-2009 \(n = 6\)). These stations were installed in habitats closely resembling actual den locations based on GPS collar data and GIS analyses. Stations for the 2007-2008 winter were retrieved in the spring of 2008 and stations for the 2008-2009 winter will be retrieved in the spring of 2009. Additional weather stations will also be installed near den sites to obtain data for the 2009-2010 and 2010-2011 winters. Once all stations are retrieved, data will be pooled and analyzed for potential relationships between denning behaviour and weather patterns.

\textbf{Habitat Selection and Weather Variables}

In the summers of 2007 and 2008, we installed seven HOBO\textsuperscript{®} Micro Stations within the main habitat types of the study area in order to consider the influence of weather variables on grizzly bear habitat selection. In 2007, permanent weather stations were installed in the main habitat types (pine, black spruce, and 0-10 regenerating stand) of the Kakwa area which is approximately in the center of the study area (\(n = 3\); Figure 1). These three weather stations will act as base stations and will be recording data in the same stands until 2011. In 2008, four other weather stations were installed in the western portion of the study area within a pine, black spruce, broadleaf, and a 0-10 year old regenerating stand (Figure 1). The spring of 2009 and 2010 respectively, these four weather stations will be re-installed in the same stand types within the northern and southern regions of the study area in order to get adequate spatial coverage.

In 2008, thirty operative temperature sensors were also installed in the main habitat types of the study area. Chosen habitat types were shrub, treed wetland, broadleaf, pine, black spruce, and regenerating stands (0-10 year old, 10-25 year old, 25-35 year old, and 35-45 year old). Three operative temperature sensors were installed per habitat type and were spread throughout the area most extensively used by grizzly bears during that year. These same operative temperature sensors will be recording data within the study area until spring 2011. Operative temperature sensors give an accurate measure of the “perceived temperature” including the effects of solar radiation, canopy cover, and wind.
These sensors will be visited periodically in order to insure proper equipment performance and download logged data.

Habitat types were classified using the Weyerhaeuser harvest data and the project’s remote sensing team 2007 deliverables.

![Figure 1: Permanent weather stations (Kakwa area), 2008 weather stations (Two Lakes area), and Operative Temperature sensors throughout the Weyerhaeuser study area 2008.](image)

**Berry plots**

Because habitat selection and movement rates are unequivocally linked to food availability and abundance, and because berries are the main grizzly bear food source in late summer and early fall, we installed 270 permanent berry plots (0.5m²) for six berry-producing shrub species recognized as important grizzly bear food in the Weyerhaeuser study area (Figure 2). Selected shrubs species were *Vaccinium membranaceum* (100 plots), *V. myrtilloides* (100 plots), *V. caespitosum* (100 plots), *V. vitis-idea* (200 plots), *Empetrum nigrum* (30 plots), and *Shepherdia Canadensis* (45 plots). All berry plots were stratified by percent cover of shrub and positioned on slopes of less than 10º within the three main berry-producing habitat types of the study area (70-100% pine, spruce (< 30% pine), and 10-25 year old regenerating stands). Berry plots were installed in July 2008 and re-visited systematically in early and late August in order to assess berry productivity. In early August, if most berries were ripe within a plot, the ripe berries were weighted for...
biomass estimates and in late August all unripe and ripe berries of leftover plots were weighted.

These permanent berry plots were built in an effort to assess yearly variations in berry availability and will therefore be re-visited in July and August of 2009 and 2010.

Lastly, in order to quantify berry availability within an individual grizzly bear’s home range, the distribution and occurrence of berry-producing shrub species will be assessed by visiting randomly picked 5-hectare plots within pine, spruce, and 10-25 year old regenerating stands of the study area. To this date, three 5-hectare plots were assessed for each of these habitat types and several more will be visited in 2009 and 2010. Within each 5-hectare plot, fifty random locations were visited. At these locations, we recorded presence/absence and percent cover of berry-producing shrub species. Once a significant number of locations are visited, an accurate measure of yearly berry biomass available within a grizzly bear’s home range will be derived from the data. This data will be analysed with weather data, GPS-collar data (habitat selection, and movement rates) as well as dates of den entry to assess the influence of weather variables on grizzly bear behaviour.

![Figure 2: Permanent Berry plots in the Weyerhaeuser study area 2008.](image-url)
CHAPTER 3: QUANTIFYING GRIZZLY BEAR FOOD IN PINE AND HARVESTED HABITATS: ASSESSING MOUNTAIN PINE BEETLE HARVESTING SCENARIOS AND IMPACTS TO GRIZZLY BEAR HABITAT

Terry Larsen\textsuperscript{1,4}

\textsuperscript{1}Foothills Research Institute; \textsuperscript{4}University of Alberta

Introduction
Grizzly bears (\textit{Ursus arctos}) are generalist omnivores utilizing seasonally available food resources that vary considerably across spatial and temporal scales (Munro et al. 2006). While this variation makes extrapolating results from one study population to another difficult (Munro et al. 2006), food resources can be grouped into categories that contribute significantly to differences in population size and vital rates (Hilderbrand et al. 1999a; Reynolds-Hogland et al. 2007). Hilderbrand et al. (1999a) found that body mass, litter size, and population density were positively correlated with the proportion of meat in a coastal grizzly bear’s diet and is attributed to lipid rich fall salmon. Interior grizzly bear populations consume terrestrial meat sources from predation or scavenging (Hilderbrand et al. 1999a); however, while the relative proportion of ungulates, predominantly neonates, can comprise a significant proportion of the spring diet (Munro et al. 2006; McLellan and Hovey 1995), mass gain during this period is in lean body mass whereas in the fall, it is in fat (Hilderbrand et al. 1999a). Fruit is thought to be an important food resource sustaining several interior grizzly bear populations (Munro et al. 2006; McLellan and Hovey 1995; Welch et al. 1997) and allows them to meet the energy demands of cub production and winter hibernation (Welch et al. 1997). While summer and fall fruit is likely a key driver of grizzly bear population dynamics because of tremendous annual variation, the availability of energy rich spring food resources such as ungulates and herbaceous items is important for females post denning to meet the energy requirements of cub rearing (Hilderbrand et al. 1999b).

Along the Eastern Slopes of the Rocky Mountains in Alberta are large tracts of mature contiguous lodgepole pine (\textit{Pinus contorta}) forests, a result of extensive fire suppression efforts. Recently, the threat of mountain pine beetle infestation reaching epidemic levels similar to British Columbia has prompted the provincial government to develop a long-term pine management strategy (Alberta Sustainable Resource Development 2006). The strategy directs forest harvesting companies to reduce susceptible pine forests by 75% over 20 years within management area boundaries (Alberta Sustainable Resource Development 2006). This unprecedented level of harvesting raises concern for grizzly bears because pine forests encompass a large proportion of current grizzly bear range. Nielsen et al. (2004) showed that forest harvesting alters the distribution of seasonally
critical grizzly bear foods, which is related to the age and type of mechanical treatment since harvesting, and fine scale environmental conditions. However, quantitative models describing relative abundance of grizzly bear food along a forest age trajectory and the influence of environmental gradients are currently lacking. Understanding how landscape level food supply and dynamics will change under proposed forest harvesting scenarios is an important component of managing habitat in support of grizzly bear conservation in Alberta.

Methods

Study Area

The study area is located within Weyerhaeuser Grande Prairie’s Forest Management Agreement (WGP; 10,000 km²; 119° 13’W and 54° 32’N) north of Grande Cache, Alberta, Canada. A variety of industrial and recreational land-use activities occur throughout WGP; however, forest harvesting is viewed as the primary disturbance mechanism (Nielsen et al 2004). Throughout Alberta, forest companies have used a two-pass forest harvesting system whereby a stand of merchantable coniferous of deciduous timber is harvested and allowed to regenerate for approximately 15-20 years before the adjacent stand is logged. This method of harvesting creates a patch work of regenerating and adjacent mature forest stands during the first pass. Approximately 15% (1500 km²) has been harvested since the early 1970’s between 900-1900m in elevation. Surrounding these regenerating harvested areas are a mosaic of forested (e.g. conifer, mixed-wood, and deciduous) and non-forested (e.g. shrub and herbaceous) habitat types.

Three natural sub-regions (Sub-alpine, Upper and Lower Foothills) encompass the majority of the study area. Elevations range from 650-2300m. Annual mean precipitation is between 550-1050mm, and annual mean temperatures are 4.7-11.3 degrees Celsius. A prominent gradient exists between higher elevations and rugged terrain in the south-west to gently rolling terrain and lower elevations to the north-east. Tree species composition transitions from conifers at higher elevation to mixed conifers and deciduous trees at lower elevations. The Lower-Subalpine region (<2000m) is dominated by closed fire-origin lodgepole pine stands on drier sites with mixed Engelmann spruce (*Picea engelmannii*) subalpine fir (*Abies lasiocarpa*) and pine on moister slopes. Fen areas (poor drainage) are comprised of willow (*Salix spp.*), black and white spruce (*Picea mariana* and *glauca*), and tamarack (*Larix laricina*). The Upper Foothills can be characterized as pure stands of lodgepole pine or a mix of pine with black and/or white spruce. Stands of white spruce are commonly found along riparian areas while mixed stands of balsam fir (*Abies balsamifera*), spruce species, and lodgepole pine are found on moister slopes. Deciduous stands of aspen (*Populus tremuloides*) and balsam poplar (*Populus balsamifera*) are less common and found only on southern and westerly aspects. Shrubby grasslands occur on drier sites with tamarack (*Larix laricina*) and black spruce fens along poorly drained areas. At higher elevations of the Lower Foothills, pure stands of pine occur with mixed pine, spruce, and deciduous stands depending on moisture regimes. At lower elevations, deciduous stands are common on all aspects.
Grizzly Bear Food Resources
We identified primary and secondary food resources associated with conifer forests and harvested areas from previous grizzly bear habitat and diet analyses conducted in west-central Alberta (Munro et al. 2006; Unpublished Foothills Research Institute). Primary foods are those that form the majority of the annual diet of grizzly bears, and are considered critical for individual growth and reproduction. These are sweet-vetch roots (*Hedysarum spp.*), horsetails (*Equisetum spp.*), clover (*Trifolium spp.*), cow parsnip (*Heracleum lanatum*), black huckleberry (*Vaccinium membranaceum*), dwarf blueberry (*Vaccinium caespitosum*), velvet-leaf blueberry (*Vaccinium myrtilloides*), and buffaloberry (*Shepherdia canadensis*). Secondary foods do not make up a significant proportion of an annual grizzly bear diet and includes graminoids (grasses and sedges), twisted stalk (*Streptopus amplexifolious*), honeysuckle (*Lonicera involucrate*), wild raspberry (*Rubus idaeus*), lingonberry (*Vaccinium vitis-idaea*), ant mounds, and ant substrate (coarse woody debris and stumps >10cm in diameter at breast height).

Habitat Stratification
We used an Alberta Vegetation Inventory (AVI) database from WGP to identify the age of pine forest and harvested habitat within the Upper Foothills natural sub-region. Pine forests were separated into five age categories (40-59, 60-89, 90-119, 120-159, and >160 years) and into seven age categories for harvested areas (1974-1978, 1979-1983, 1984-1988, 1989-1993, 1994-1998, 1999-2003, and 2004-2007). Pine forests were further broken down into two classes of percent pine composition at the stand level (50-70 and 80-100). Points were randomly generated (1 point location within a minimum patch size of 900m$^2$; point density/patch=0.0001234) across the pine and harvested strata. Due to logistical constraints and because pine forests are widespread throughout the study area often in areas with no road access, we selected random locations within buffered distances as follows: major gravel roads 700m; secondary gravel 500m; and truck trails 200m. Because harvested areas were generally more accessible and made up a smaller proportion of the area, we considered all randomly generated locations within harvested areas as part of the random selection process of locations to visit. A Compound Topographic Index (CTI) was used to match moisture conditions within pine forests and harvested areas; random sampling occurred along this environmental gradient.

Sampling Strategy
Random locations were visited from June 15 to September 1 2008. At each location a 30 x 30 meter sample plot was established with the perimeter of the plot at a minimum of 30 meters from any anthropogenic edge such as a seismic line, well-site, harvested block or retention patch within a harvested block (Figure 1). Within each sample plot, 5 – 1 x 30 meter transects, orientated south to north at the 0.5, 7.5, 15, 22.5, and 29.5 meter marks (from west to east) and 5 – 1 x 1 meter subplots at the same distances along each transect were used to quantify primary and secondary foods. Percent cover was occularly estimated for those food items that could not be efficiently counted because of their growth patterns. Stems, individual plants, or individual items were counted for all other food resources.
Results

Food occurrence

Vegetation plots in pine forests (n=70) and harvested habitat (n=70) revealed that most food items were available in both the habitat types. Clover and sweet-vetch were absent from pine forests and ant mounds were not found in any of the habitat types.
Relative food abundance
Considerable differences were observed between the relative abundance of food resources in pine and harvested habitat (Table 1.). This includes differences among plant percent cover or stem counts and the amount of fruit between habitat types. Primary fruit producing shrub relative abundance was higher in pine forests except for Velvet-Leaved Blueberry. However, there was more Dwarf Bilberry fruit in harvested areas than in pine with a similar result for Buffaloberry. In 6 of 8 food species, fruit production was higher in harvested areas than in pine forest habitat. There were substantially more Black Huckleberries in pine forest than harvested areas. Cow parsnip was considerably higher in harvested areas than in pine forest. Primary herbaceous items such as horsetails and cow parsnip were found in greater relative abundance in harvested habitat. Secondary fruit producing shrubs displayed similar trends as did primary foods with more fruit in harvested habitat. Similar relative abundance of substrate (coarse woody debris and stumps) was found in pine and harvested habitat.

Table 1: Mean and standard error of percent cover, counts of stems and fruit in pine forest (>=40 years; n=70) and harvested (<=35 years; n=70) habitat in the Upper Foothills of west-central Alberta, Canada. Means are the average of 5 transects (1x30 meters) among sample plots and habitat types.

<table>
<thead>
<tr>
<th>Quantification Method</th>
<th>Food Resource</th>
<th>Harvested Mean</th>
<th>Harvested SE</th>
<th>Pine Forest Mean</th>
<th>Pine Forest SE</th>
<th>Harvested Fruit Counts Mean</th>
<th>Harvested Fruit Counts SE</th>
<th>Pine Forest Fruit Counts Mean</th>
<th>Pine Forest Fruit Counts SE</th>
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<td>Subplots</td>
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<td>0.2</td>
<td>1.1</td>
<td>0.2</td>
<td>31.6</td>
<td>14.4</td>
<td>4.3</td>
<td>2.0</td>
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<td></td>
<td>Black Huckleberry</td>
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<td>0.5</td>
<td>4.6</td>
<td>0.7</td>
<td>5.8</td>
<td>2.6</td>
<td>32.8</td>
<td>8.7</td>
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<td>Velvet-Leaved Blueberry</td>
<td>3.2</td>
<td>0.9</td>
<td>2.6</td>
<td>0.5</td>
<td>138.6</td>
<td>39.6</td>
<td>17.8</td>
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</tr>
<tr>
<td></td>
<td>Lingonberry</td>
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<td>2.8</td>
<td>0.4</td>
<td>22.5</td>
<td>10.1</td>
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<td></td>
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<td>33.9</td>
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<td>Graminoids</td>
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<td>-</td>
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<td></td>
<td>Cow Parsnip</td>
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<td>5.3</td>
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<td>0.8</td>
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<td>-</td>
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<td>-</td>
</tr>
<tr>
<td></td>
<td>Twisted Stalk</td>
<td>4.3</td>
<td>1.4</td>
<td>5.3</td>
<td>1.4</td>
<td>-</td>
<td>-</td>
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<td>-</td>
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<td></td>
<td>Coarse Woody Debris</td>
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<td>18.7</td>
<td>2.4</td>
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<td>-</td>
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</tr>
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<td></td>
<td>Stumps</td>
<td>10.9</td>
<td>1.1</td>
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<td>-</td>
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<td>Ant Mounds</td>
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<td>0.0</td>
<td>0.0</td>
<td>-</td>
<td>-</td>
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</tr>
</tbody>
</table>

Food abundance and age
Age influenced the mean percent cover and counts of primary and secondary food resources in pine forest and harvested habitats. Velvet-Leaved Blueberry peaked between the ages of 16-20 years while Black Huckleberry peaked between 90-120 years (Figure 2.). Horsetails became abundant from ages 6 through to 20 and then declined.
steadily as age increased (Figure 3.). Cow parsnip abundance was relatively high from 20-35 years of age and peaked between 31 to 35 years. Shortly after harvesting, Raspberries were highly abundant and then declined sharply at 6 years of age (Figure 4). Coarse woody debris was most abundant shortly post harvest for about ten years and then declined around 25 years of age with a sharp increase in 60 year old pine stands (Figure 5).

Figure 2. Mean percent cover of fruit producing shrubs along an age gradient in the Upper Foothills of west-central Alberta, Canada. Age categories are represented by pine forest (>=40 years) and harvested (<=35 years) habitat.
Figure 3. Mean counts of individual horsetails along an age gradient in the Upper Foothills of west-central Alberta, Canada. Age categories are represented by pine forest (≥40 years) and harvested (≤35 years) and habitat.

Figure 4. Mean stem counts of herbaceous and fruit producing shrubs along an age gradient in the Upper Foothills of west-central Alberta, Canada. Age categories are represented by pine forest (≥40 years) and harvested (≤35 years) habitat. Raspberry is scaled by the primary (left) y-axis only.
Figure 5. Mean counts of stumps and coarse wood debris along an age gradient in the Upper Foothills of west-central Alberta, Canada. Age categories are represented by pine forest (>=40 years) and harvested (<=35 years) habitat.

Discussion
Comparing these results to previous habitat studies that investigated the occurrence of commonly used food resources by grizzly bear in the foothills of west-central Alberta was both similar and strikingly different. Two primary food items, Sweet Vetch and Buffaloberry, considered critical foods for grizzly bears during the spring, summer, and fall were practically absent or had a low frequency of occurrence (Munro et al. 2006). Because these species are commonly found in the foothills of Alberta, environmental conditions (i.e. soils and/or moisture) could be playing a role in their observed distribution further north in this study area. Other differences included the lack of clover and abundance of cow parsnip in this study area as opposed to the foothills further south (Nielsen et al. 2004). Some similarities were also evident between these areas with coniferous forests containing more Black Huckleberries, Ligonberry, and Dwarf Bilberry and harvested habitats had more Horsetails (Nielsen et al. 2004). Fruit production was also similar among these areas with harvested areas containing more except for Black Huckleberries and Ligonberry (Nielsen et al. 2004). Coarse woody debris substrate is known to contain ants that are used by grizzly bears (Nielsen et al. 2004), it is not known whether bears use this substrate in forests for ant foraging. Stump substrate found only in harvested habitat is known to contain ants that are utilized by grizzly bears (Nielsen et al. 2004).

Although these results are preliminary, primary foods both herbaceous and fruit were more abundant among mid-seral harvested areas than in pine forest habitat. However, pine forests do contain large quantities of Black Huckleberries, which are an important food source for other bear populations in North America (McLellan and Hovey 1995). Because Velvet-Leaved Blueberries are quite small relative to Black Huckleberries, size relative to energetic value should be considered to properly evaluate the value of one versus the other for grizzly bears. Furthermore, the availability of these two important
food resources may also differ temporally. Future modeling and statistical analysis will consider the effect of different environmental gradients known to influence grizzly bear food distribution and abundance.

Literature Cited
CHAPTER 4: REMOTE SENSING MAPPING AND RESEARCH UPDATE (U of C, U of S)

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INTRODUCTION AND OVERVIEW

2008/09 marks the ninth year in which researchers and technicians from the University of Calgary and the University of Saskatchewan have contributed towards remote sensing research and mapping initiatives in the Foothills Research Institute Grizzly Bear Program (FRIGBP). Activities in the past fiscal year can be divided broadly into two main categories: (i) those related to deliverable products, generated primarily by technical staff, and (ii) those related to research pursuits, performed primarily by graduate students and research personnel. In this report, we summarize the activities taking place within the Foothills Facility for Remote Sensing and GIScience at the University of Calgary and the Environmental Remote Sensing Laboratory at the University of Saskatchewan. On the deliverables front, the remote sensing/mapping team completed two significant milestones:

1. Annual changes in landcover, vegetation, and landscape structure in the Grande Cache unit from 2005-2008, and
2. A spatially-explicit map of Pinus across the Grande Cache unit

On the research front, efforts this year were concentrated in three broad research topics that support the development of knowledge and planning tools that make up the Grizzly Bear Program’s core goals:

3. Landscape monitoring and change detection,
4. Vegetation dynamics and phenology, and
5. Remote sensing and habitat mapping.

Summary information on the status and achievements in each of the five topic areas are contained in the following sections.
Increasing levels of human activity in west-central Alberta surrounding the timber, coal, and petroleum industries have lead to widespread growth in anthropogenic disturbance features on the landscape, and corresponding changes in land cover structure and composition. A primary goal of the mapping team in this project has involved the accurate detection, mapping, and quantification of human-induced landscape changes at a temporal resolution matching our grizzly bear health and stress data sets. We employed satellite imagery and change-detection procedures to generate spatially-explicit layers of land cover, vegetation, and various anthropogenic disturbance features across the Grande Cache unit (Figure 1), then quantified their annual levels and change trajectories using a variety of summary statistics.

Figure 1: Location of the Grande Cache unit study area in west-central Alberta.
Methods
The methods used to create annual map products of the Grande Cache unit followed the disturbance inventory approach to landscape monitoring described fully in Linke et al. (2009a). While currently in press at the time of writing, the paper is scheduled for publication in June of 2009, and represents an important (and exhaustive) description of our methods, which we summarize only briefly here.

We acquired a series of Landsat Thematic Mapper and Enhanced Thematic Mapper Plus imagery (Table 1) to track annual change patterns across the Grande Cache unit from 2005 to 2008. This information was supplemented by SPOT orthomosaics and airborne orthophotography obtained through data-sharing agreements with the provincial government and other program partners. Disturbance features were mapped primarily through manual digitizing and update of available GIS layers. Annual disturbance features were visually inspected and manually corrected where necessary, and subjected to boundary conditioning routines designed to reduce the occurrence of slivers (Linke et al. 2009b). Disturbance objects were transformed to land cover classes using decision rules (e.g. road features = barren; pipelines = herbaceous), and spatially mosaicked to create annual update layers. Finally, we overlaid the annual update layers on the co-registered 2004 base land cover map to update land cover products for each year of interest.

Table 1: Landsat scenes acquired and processed for quantifying human disturbance, land cover, and landscape structure from 2005 to 2008.

<table>
<thead>
<tr>
<th>Landsat Path/Row</th>
<th>Image Acquisition Date</th>
<th>Sensor</th>
</tr>
</thead>
<tbody>
<tr>
<td>46/22</td>
<td>August 11, 2004</td>
<td>Landsat 7 ETM+</td>
</tr>
<tr>
<td></td>
<td>September 2, 2006</td>
<td>Landsat 5 TM</td>
</tr>
<tr>
<td></td>
<td>August 8, 2008</td>
<td>Landsat 5 TM</td>
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</table>

Results and Discussion
Base Map Products
The results of our disturbance inventory-update procedures was a series of land cover, crown closure, and tree species composition maps covering the time period from 2004 to 2008 (Figure 2). While an accuracy assessment of the maps was not completed at this stage, we anticipate the quality to be similar to that noted in BMAs 3 & 4 further south, wherein the accuracy of disturbance identification (100% overall accuracy; Kappa=1.0) and labeling (93% overall accuracy; Kappa=0.889) was very good. It is important to note, however, that the overall quality of the maps in general depends also on accuracy of the original base maps.
Figure 2: Samples of changing land cover (top), crown closure (middle), and tree species composition (bottom) over a small portion (overview) of the Grande Cache unit, 2004 to 2008.
**Human Disturbance Features**

A summary of the amounts, densities, and proportions of key human disturbance features across the Grande Cache unit for the time period 2004 to 2008 can be found in Table 2. These same trends are shown graphically in Figure 3.

For each year, we documented substantial increases in human disturbance features, clearly reflecting the growing impact of fast-developing resource extraction industries in west-central Alberta grizzly bear habitat. Road density was observed to increase from 0.44 km/km² in 2004 to 0.51 km/km² in 2008. Cutblock proportion increased from 7.9% in 2004 to 9.6% in 2008. Wellsite density changed from 0.18/km² in 2004 to 0.26/km² in 2008. Pipeline density was observed to increase from 0.15 km/km² in 2004 to 0.17 km/km² in 2008.

**Table 2: A summary of road density, cutblock proportion, wellsite density, and pipeline density, across the Grande Cache unit from 2004 to 2008.**

<table>
<thead>
<tr>
<th>Year</th>
<th>Road Density (km/km²)</th>
<th>Cutblock Proportion (%)</th>
<th>Wellsite Density (#/km²)</th>
<th>Pipeline Density (km/km²)</th>
</tr>
</thead>
<tbody>
<tr>
<td>2004</td>
<td>0.44</td>
<td>7.9</td>
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<td>0.15</td>
</tr>
<tr>
<td>2005</td>
<td>0.45</td>
<td>8.1</td>
<td>0.19</td>
<td>0.15</td>
</tr>
<tr>
<td>2006</td>
<td>0.49</td>
<td>9.0</td>
<td>0.24</td>
<td>0.16</td>
</tr>
<tr>
<td>2007</td>
<td>0.50</td>
<td>9.4</td>
<td>0.25</td>
<td>0.16</td>
</tr>
<tr>
<td>2008</td>
<td>0.51</td>
<td>9.6</td>
<td>0.26</td>
<td>0.17</td>
</tr>
</tbody>
</table>
Figure 3: Trends in road density, cutblock proportion, wellsite density, and pipeline density across the Grande Cache unit from 2004 to 2008.
Summary
We employed the disturbance-inventory framework of Linke et al. (2009a) to generate spatially-explicit layers of land cover, vegetation, and various anthropogenic disturbance features across the Grande Cache unit. When combined with previous annual mapping efforts in BMAs 3 and 4 to the south, these products form the basis of a multi-year landscape pattern analysis documenting the structural changes in grizzly bear habitat across its core range in west-central Alberta (Figure 4).

TOPIC 2: PINE DISTRIBUTION OF OCCURRENCE MAP

Previous map products of tree species composition produced by the remote sensing team have been limited to continuous estimates (0-100%) of the proportion of coniferous tree species occurring within a 30-metre pixel. While certainly suitable for basic characterization of tree species composition and general habitat-mapping, recent focus on issues related to mountain pine beetle research in the Grande Cache unit has created a need for more detailed characterization of Pinus.

The pine map represents the distribution of pine occurrence (presence-absence), as predicted by remote sensing variables using a classification tree analysis technique. Classification trees are used to predict membership in categories of a dependent variable from their measurements on one or more predictor variables. In this case, the classification tree program, See5, was used to create decision rules regarding the occurrence of pine (dependent variable), based on spectral information from Landsat TM imagery (path 46, row 22) at specific locations where pine presence and pine absence were known. In the analysis, 198 pine presence points were used along with 1,980 absence points, for 2178 total points and a ten-to-one absence-to-presence ratio. Presence and absence points were obtained via field visits and absence points were supplemented with points collected from existing GIS data sources. Presence points were identified as such if greater than or equal to one pine tree was identified. The decision rules were extracted from See5 and implemented using the Raster Calculator in ArcMap to create the final layer of pine occurrence (Figure 5).

![Figure 5: Presence and absence of Pinus within the Grande Cache unit (Landsat path 46, row 22).]
TOPIC 3: LANDSCAPE MONITORING AND CHANGE DETECTION

3-1) Object-based approaches to change analysis and thematic map update: challenges and limitations

Bi-temporal change analysis strategies performed in an object-oriented environment are prone to the generation of sliver objects: small, spurious polygons created by the inconsistent delineation of persistent change features appearing in consecutive co-registered images. The issue represents a methodological challenge that can seriously limit the visual and structural quality of the finished map product if not adequately addressed. A critical analysis of annual landcover maps generated by updating and backdating object-oriented reference maps in a western Alberta study area revealed that sliver objects made up between 3% and 12% of the total change features identified by the routine, despite high thematic accuracies. The results highlight the emerging need for a methodological framework designed to handle the spatial challenges posed by change analysis in an object-based environment.


3-2) The influence of patch delineation mismatches on multi-temporal landscape pattern analysis

Investigations of land-cover change often employ metrics designed to quantify changes in landscape structure through time, using analyses of land cover maps derived from the classification of remote sensing images from two or more time periods. Unfortunately, the validity of these landscape pattern analyses (LPA) can be compromised by the presence of spurious change, i.e., differences between map products caused by classification error rather than real changes on the ground. To reduce this problem, multi-temporal time series of land-cover maps can be constructed by updating (projecting forward in time) and backdating (projecting backward in time) an existing reference map, wherein regions of change are delineated through bi-temporal change analysis and overlaid on to the reference map. However, this procedure itself creates challenges, because sliver patches can occur in cases where the boundaries of the change regions do not exactly match the land-cover patches in the reference map. In this paper, we describe how sliver patches can inadvertently be created through the backdating and updating of land-cover maps, and document their impact on the magnitude and trajectory of four popular landscape metrics: number of patches (NP), edge density (ED), mean patch size (MPS), and mean shape index (MSI). In our findings, sliver patches led to significant distortions in both the value and temporal behaviour of metrics. In backdated maps, these distortions caused metric trajectories to appear more conservative, suggesting lower rates of change for ED and inverse trajectories for NP, MPS and MSI. In updated maps, slivers caused metric trajectories to appear more extreme and exaggerated, suggesting higher rates of change for all four metrics. Our research underscores the need to eliminate sliver patches from any study dealing with multi-temporal LPA.

3-3) A disturbance-inventory framework for flexible and reliable landscape monitoring.
Remote sensing plays a key role in landscape monitoring, but our handling of these data in a multi-temporal time series is not yet fully developed. Of particular concern is the presence of spatial and thematic errors in independently created maps that distort measures of landscape pattern and constrain the reliability of change analysis. In addition, there is a need to incorporate continuous attributes of cover gradients for flexible map representations that support a variety of applications. In this paper, we present a framework for generating temporally and categorically dynamic land-cover maps that provide such a reliable and adaptable foundation. The centerpiece is a spatio-temporal disturbance-inventory database, created through semi-automated change detection and conditioned with boundary-matching procedures, which can be used to backdate and update both continuous and categorical reference maps. We demonstrate our approach using multi-annual Landsat imagery from a forested region in west-central Alberta, Canada, between the years 1998 and 2005.


3-4) MODIS-based change detection for grizzly bear habitat mapping in Alberta
Coarse resolution data from the Moderate Resolution Imaging Spectroradiometer (MODIS) was used to test the effectiveness of 250 m data to detect forest disturbances and update an existing, large-area (150,000 km²), 30 m Landsat ETM+ and TM land-cover map product used in Grizzly Bear (Ursus arctos) habitat analysis. A Landsat-derived polygon layer was applied to the MOD13Q1 data product to create a polygon-based, mean NDVI time series (2000 to 2005). Image differencing of the dataset produced multiple-scale layers of change including a two-date, five-year change and a five year composite of annual changes. Accuracy assessments based on available GIS data showed an overall accuracy as high as 59 percent. Results also show that disturbance patches larger than 15 ha were represented with an accuracy of 75 percent or higher. This offers an alternative to higher spatial resolution data for the identification of larger features and also provides general change information for those areas that may be suitable for analysis with higher spatial resolution data.

3-5) Using SPOT and ASTER imagery to update large-area Landsat-based map products in Alberta’s Rocky Mountains

In Alberta, several large-area mapping programs rely on Landsat data for base mapping products, including general landcover and detailed forest structure attributes (e.g., leaf area index, crown closure). Frequent updates to these products are required to detect changes caused by forest harvesting and other human and natural disturbances. Given the undetermined future status of Landsat data availability, alternatives to Landsat imagery must be tested; two possible candidates for alternative update data are imagery acquired by i) the Advanced Spaceborne Thermal Emission and Reflectance Radiometer (ASTER) and ii) the Satellite Pour l’Observation de la Terre (SPOT) sensors. For this study, existing map products derived from Landsat data were compared to forest landcover and crown closure maps generated from ASTER and SPOT data. Using the same coverage area, training sets, and classification method, an overall accuracy of 86.6% for ASTER landcover classification, and an overall accuracy of 93.7% for SPOT landcover classification, were obtained in an independent sample of field-checked sites. The ASTER and SPOT classification accuracies were higher than the Landsat-based landcover classification accuracy, with most of the differences apparent in the shrub and herb classes. Some small disturbance features (e.g., well-sites, access roads) appeared to be more accurately detected using the finer-spatial resolution SPOT image data. The crown closure maps derived from ASTER and SPOT data represented the spatial distribution of the vegetation well, with the overall accuracy of the crown closure maps calculated to be over 88%. This is a small increase in accuracy when compared to the results obtained using the Landsat imagery. Spatial differences in the final maps, expressed in the form of landscape metrics, were minor although visual analysis revealed potential problems requiring post-processing refinements. Overall, these results demonstrate that in these environments, ASTER and SPOT can be used to map landcover and crown closure with accuracy comparable to that obtained using Landsat data, providing an alternative data source for map update in future.


**TOPIC 4: PHENOLOGY AND VEGETATION DYNAMICS**

4-1) Noise reduction of NDVI time series: An empirical comparison of selected techniques

Satellite-derived NDVI time series are fundamental to the remote sensing of vegetation phenology, but their application is hindered by prevalent noise resulting chiefly from varying atmospheric conditions and sun-sensor-surface viewing geometries. A model-based empirical comparison of six selected NDVI time series noise-reduction techniques revealed the general superiority of the double logistic and asymmetric Gaussian function-fitting methods over four alternative filtering techniques. However, further analysis demonstrated the strong influence of noise level, strength, and bias, and the extraction of phenological variables on technique performance. Users are strongly cautioned to consider both their ultimate objectives and the nature of the noise present in an NDVI
data set when selecting an approach to noise reduction, particularly when deriving phenological variables.


### 4-2) Processing NDVI time series for characterizing vegetation phenology: The case against noise reduction

We conducted a series of model-based empirical analyses in order to assess the value of six selected processing techniques designed to reduce the effects of persistent noise in satellite-derived time series of the normalized difference vegetation index (NDVI): an imperfectly understood family of routines whose value has rarely been questioned in the literature. Our results showed that while each of the six techniques was generally beneficial for reducing unwanted high-frequency variations in the data, the procedures tended to distort the phenological integrity (e.g. the shape and amplitude) of the time series, to the point where the accuracy of derived phenologically-based metrics was notably decreased. This observation remained consistent over the range of conditions we examined, implying a level of robustness to the trend. The choice of phenological metric was shown to exert a strong influence on the potential benefits of noise reduction, along with the level of noise, and, to some extent, biogeographical region. Users of these techniques are strongly encouraged to exercise caution when the ultimate objective is extraction of phenologically-based metrics (start of season, end of season, etc), and are advised against the use of noise-reduction on relatively clean time series where noise levels in the data are minimal.

**TOPIC 5: REMOTE SENSING AND HABITAT MAPPING**

5-1) Remote sensing and forest inventory for wildlife habitat assessment
Researchers and managers undertaking wildlife habitat assessments commonly require spatially-explicit environmental map layers such as those derived from forest inventory and remote sensing. However, end users of geospatial products must often make choices regarding the source and level of detail required for characterizing habitat elements, with few published resources available for guidance. We appraised three environmental data sources that represent options often available to researchers and managers in wildlife ecological studies: (i) a pre-existing forest inventory; (ii) a general-purpose, single-attribute remote sensing land cover map; and (iii) a specific-purpose, multi-attribute remote sensing database. The three information sources were evaluated with two complementary analyses: the first designed to appraise levels of map quality (assessed on the basis of accuracy, vagueness, completion, consistency, level of measurement, and detail) and the second designed to assess their relative capacity to explain patterns of grizzly bear (*Ursus arctos*) telemetry locations across a 100,000-km² study area in west-central Alberta, Canada. We found the forest inventory database to be reasonably functional in its ability to support resource selection analysis in regions where coverage was available, but overall, the data suffered from quality issues related to availability, accuracy, and consistency. The general-purpose remote sensing land cover product ranked higher in terms of overall map quality, but demonstrated a lower capacity for explaining observed patterns of grizzly bear habitat use. We found the best results using the specific-purpose, multi-attribute remote sensing database, and recommend that similar information sources be used as the foundation for wildlife habitat studies whenever possible, particularly those involving large areas that span jurisdictional boundaries.


5-2) Mapping the distribution of whitebark pine (*Pinus albicaulis*) in Waterton Lakes National Park using logistic regression and classification tree analysis
Accurate spatial information on the distribution of whitebark pine – a keystone species in alpine environments across western Canada – is critical for the planning of conservation activities designed to ameliorate the damaging effects of blister rust, mountain pine beetle, and interspecific competition. We compared classification tree analysis and logistic regression analysis to explore their relative abilities to model whitebark pine presence and absence with medium-spatial-resolution satellite and topographic variables across a complex study site in Waterton Lakes National Park, Alberta. Both techniques were found to be effective, generating map products of roughly equal thematic quality (91% overall accuracy; kappa=0.76). However, the logistic model was valuable in its ability to predict ratio-level probability surfaces, while the classification tree approach was simpler, faster, and found to generate a slightly more balanced model from an
individual class accuracy perspective. End users selecting between the two techniques should make choices that balance flexibility with simplicity, while always taking care to exercise sound modeling practices.


5-3) Processing discrete-return profiling LiDAR data to estimate canopy closure for large-area forest mapping and management

We performed a series of empirical experiments designed to refine the processing of discrete-return profiling LiDAR data for the purpose of estimating canopy closure across a broad range of forest conditions in west-central Alberta, Canada. The following three methodological conclusions were obtained: (i) a new line segment method based on the ratio of vegetation segment distance over total distance outperformed alternative point-count techniques described previously in the literature; (ii) an absolute canopy-ground threshold of 1.4 m generated the best models overall, and appeared to extend well across a range of forest types; and (iii) stratification by species composition (hardwood, softwood, and mixedwood) or moisture regime (upland and wetland) was of little influence in alternate models, suggesting good portability of these methods across a broad variety of forest conditions. A \( k = n \) cross validation approach produced an average RMSE of 7.2% for the best model. In addition to contributing towards the identification of sound methodological practices, these results successfully reconciled the conceptual differences between canopy closure, which is measured through the use of ground-based optical tools, and canopy cover: captured remotely with LiDAR; revealing a direct linear relationship between the two attributes.


5-4) A medium-resolution remote sensing classification of agricultural areas in Alberta grizzly bear habitat

Habitat loss and human-caused mortality are the most serious threats facing grizzly bear (Ursus arctos L.) populations in Alberta, with conflicts between people and bears in agricultural areas being especially important. However, the agricultural land as one class in the current classified map limited the understanding of the bear habitat in agriculture region. The objectives of this research were to find the best possible classification approach from a limited selection of methods for determining multiple classes of agricultural and herbaceous land cover, and to create land cover maps of agricultural and herbaceous areas which will be integrated into existing grizzly bear habitat maps for western Alberta. Three different object-based classification methods, one unsupervised and two supervised methods, were analyzed with these data to determine the most accurate and useful method. The best method was the Supervised Sequential Masking (SSM) technique, which gave an overall accuracy of 88% and a Kappa Index of Agreement (KIA) of 83%. When combined with bear GPS location data, it was discovered that bears in agricultural areas were found in Grass / Forage crops 77% of the
time, with Small Grains and Bare Soil / Fallow fields making up the rest of the visited land-cover. The bears were found in these areas primarily in the summer months.


5-5) **Relationships between landscape structure and grizzly bear abundance and presence/absence in agricultural areas in Alberta, Canada**

Management plans to reduce problem bear conflicts in agricultural areas are seen as one of the strategies with the greatest potential to mitigate human-induced harmful effects on grizzly bear (*Ursus arctos*) populations in Alberta. The purpose of this research was to determine which, if any, landscape configurational and compositional metrics are related to grizzly bear presence or abundance in an agriculture-dominated landscape. Locational data for eight bears were examined in an area southwest of Calgary, Alberta. The 4494 km$^2$ study area was divided into 107 sub-landscapes of 42 km$^2$. Five-meter spatial resolution Indian Remote Sensing (IRS) panchromatic imagery was used to classify the area and derive compositional and configurational metrics for each sub-landscape. It was found that the amount of agricultural land did not explain grizzly bear use; however, secondary effects of agriculture on landscape configuration did. High landscape patch density and variation in distances between neighboring similar patch types were seen as the most significant metrics in the abundance models; higher variation in patch shape, greater contiguity between patches, and lower average distances between neighboring similar patches were the most consistently significant predictors in the bear presence / absence models. Grizzly bears appeared to prefer areas that were structurally correlated to natural areas, and avoided areas that were structurally correlated to agricultural areas. Grizzly bear presence could be predicted in a particular sub-landscape with 87% accuracy using a logistic regression model. Between 30% and 35% of the grizzlies’ landscape scale habitat selection was explained using these models.

2008-2009 PUBLICATIONS, REMOTE SENSING GROUP

Refereed Journal Articles


Journal Articles Pending


Book Chapters

Non-Referenced Contributions


Hilker, T., N.C. Coops, M.A. Wulder, G.J. McDermid, and J.C. White, 2008: Image fusion to produce vegetation and change information with Landsat spatial and MODIS temporal characteristics. Fall 2008 American Geophysical Union Meeting, San Francisco, California, USA.


Theses


Sanii, S., 2008: Assessing the effect of point density and terrain complexity on the quality of LiDAR-derived DEMs in multiple resolutions. MGIS Thesis, Department of Geography, University of Calgary.
**Thesis Proposals**


CHAPTER 5: REMOTE SENSING (UBC, UVic, & CFS)

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EXECUTIVE SUMMARY

This section reports upon the progress made in the 2008/09 fiscal year by the groups located at the University of British Columbia, the Canadian Forest Service, and the University of Victoria as part of the Grizzly Bear Program which was initiated by the Foothills Research Institute in Hinton, Alberta, Canada. In 2008/09, our research has focused on four main areas, the improvement of temporal resolution of base map templates (Topic 1), the monitoring of Mountain Pine Beetle attack and mitigation activities (Topic 2), the collection of ground validation data (Topic 3), and the projection of landscape recovery and change (Topic 4).

Mountain Pine Beetle (MPB) is impacting the pine land base, with MPB related harvesting and mitigation is causing significant landscape disturbance. As a result, there is a need to map these landscape disturbances at fine spatial and temporal scales. First, seasonal changes in vegetation were mapped at high temporal and spatial resolution, by combining high spatial resolution Landsat data with high temporal resolution MODIS observations. Second, a new data fusion model was developed to facilitate mapping of stand replacing disturbances, such has harvesting activities at a 25 m spatial resolution and at eight day time steps. The results from this model were then used to attribute disturbance features and classify them into different categories (such as harvesting or mining activities or fires). Additionally, edge features were developed from the generated disturbance maps in order to define the transition zones between differing land cover or functional vegetation classes. The impact of the understory phenology on remote sensing observations is currently investigated using a phenological camera network.

Mountain pine beetle attack and mitigation strategies were observed using a sequence of Landsat images, from which key parameters were extracted including the presence of beetle disturbance and timing of stand decline. The accuracy of discriminating beetle attack from healthy forest stands was assessed both spatially and temporally using three years of aerial survey data. Additionally a reconnaissance of six potential sites for monitoring vegetation phenology using time-lapse photography was undertaken during August of 2008. The sites range in elevation from approximately 1,000 to 2,000 m above sea level, and are located in the lower and upper foothills, and the sub-alpine natural sub-regions of Alberta.
TOPIC 1: IMPROVING THE TEMPORAL RESOLUTION OF BASE MAP TEMPLATES:

1-1) Developing Methods To Extract Base Products From Synthetic Landsat Images
Landsat imagery with a 30 m spatial resolution is well suited for characterizing landscape-level forest structure, disturbance and dynamics. While Landsat images have advantageous spatial and spectral characteristics for describing vegetation properties, the Landsat sensor's revisit rate, or the temporal resolution of the data, is 16 days. When considering that cloud cover may impact any given acquisition, this lengthy revisit rate often results in a dearth of imagery for a desired time interval (e.g., month, growing season, or year) especially for areas at higher latitudes with shorter growing seasons. As a result, this spatial resolution is often insufficient for particular ecological modeling and disturbance mapping exercises. In contrast to Landsat, MODIS (MODerate-resolution Imaging Spectroradiometer) has a high temporal resolution, covering the Earth up to multiple times per day, and depending on the spectral characteristics of interest, MODIS data have spatial resolutions of 250 m, 500 m, and 1000 m. By combining Landsat and MODIS data, we are able to capitalize on the spatial detail of Landsat and the temporal regularity of MODIS acquisitions.

Two different strategies are pursued in this study in order to (1) facilitate mapping of seasonal and other gradual changes in vegetation cover and (2) allow accurate determination of stand replacing disturbances such as induced by harvesting activities and fires. Determination of gradual changes in vegetation cover, phenology and seasonality is based on a published algorithm termed Spatial and Temporal Adaptive Reflectance Fusion Model (STARFM) (Gao et al., 2006). The data fusion model is applied over the study area and compared to existing satellite observations in order to assess the ability to map changes in vegetation. Mapping of anthropogenic and other stand replacing disturbance events is based on a newly developed algorithm, the Spatial Temporal Adaptive Algorithm for mapping Reflectance Change (STAARCH). In the following two sections, both algorithms are described in detail as is the current status of these sub-projects.

1-2) Methodology for production of MODIS/Landsat data blending for within year reflectance products
Introduction
Vegetation canopy biophysical and structural information are required inputs to many landscape-scale models, including ecosystem process and wildlife habitat models (Peddle et al., 1999; Sellers 1985b; Sellers et al., 1996; Townshend and Justice 1995). Since the launch of the first satellite sensors in the 1970’s, remote sensing has emerged as a key technology for providing modeling inputs in a spatially continuous fashion, with considerable progress being made in the determination of biophysical plant properties from optical sensors (Curran et al. 1992; Hall et al. 1991a; Prince 1991; Prince and Goward 1995; Tucker 1979). Key challenges are still imposed by technological
limitations, requiring trade-offs to be made between the spatial, spectral, and temporal
resolutions of an instrument, and often preventing an adequate description of ecosystem
dynamics and disturbance for modeling purposes. For instance, high spatial resolution
typically results in a smaller image footprint, or spatial extent, thereby increasing the time
it takes a satellite to revisit the same location on Earth (Coops et al. 2006). Conversely,
high temporal resolution sensors have a more frequent revisit rate and produce wide-area
coverage with a lower spatial resolution (Holben 1986; Justice et al. 1985). Landsat
imagery with a 30 m spatial resolution is well suited for characterizing landscape-level
forest structure and dynamics. While Landsat images have advantageous spatial and
spectral characteristics for describing vegetation properties, the Landsat sensor's revisit
rate, or the temporal resolution of the data, is 16 days. When considering that cloud
cover may impact any given acquisition, this lengthy revisit rate often results in a dearth
of imagery for a desired time interval (e.g., month, growing season, or year) especially
for areas at higher latitudes with shorter growing seasons. In contrast, MODIS
(MODErate-resolution Imaging Spectroradiometer) has a high temporal resolution,
covering the Earth up to multiple times per day, and depending on the spectral
characteristics of interest, MODIS data have spatial resolutions of 250 m, 500 m, and
1000 m.

Data fusion between high spatial and high temporal resolution data
In the study presented herein, we build upon the work of Gao et al. (2006) and investigate
the suitability of an existing data fusion algorithm (named STARFM) for generating
synthetic Landsat images that may then be used to investigate vegetation dynamics in
different land cover types. We assess the quality of the synthetic (predicted) Landsat
reflectance values for a number of broad vegetation classes (mainly coniferous forest), by
comparing these predictions with reflectance values from real (observed) Landsat images
acquired throughout one growing season. STARFM predicts pixel values based upon a
spatially weighted difference computed between the Landsat and the MODIS scenes
acquired at a base date (date at which a Landsat and a MODIS scenes exists) and a
prediction date for which a synthetic Landsat-like image is produced. A moving window
technique is used to minimize the effect of pixel outliers thereby predicting changes of
the center pixel using the spatially and spectrally weighted mean difference of pixels
within the window area (Gao et al., 2006). By combining Landsat and MODIS data, we
are able to capitalize on the spatial detail of Landsat and the temporal regularity of
MODIS acquisitions.

Results
We compared, on a channel-by-channel basis, the surface reflectance values (stratified by
broad land cover types) of four real Landsat images with the corresponding closest date
of synthetic Landsat imagery, and found no significant difference between real
(observed) and synthetic (predicted) reflectance values (mean difference in reflectance:
mixedwood $\bar{x} = 0.086, \sigma = 0.088$, broadleaf $\bar{x} = 0.019, \sigma = 0.079$, coniferous
$\bar{x} = 0.039, \sigma = 0.093$) (Figure 1, Figure 2). Similarly, a pixel based analysis shows that
predicted and observed reflectance values for the four Landsat dates were closely related
(mean $r^2=0.76$ for the NIR band; $r^2=0.54$ for the red band; $p<0.01$). Investigating the
trend in NDVI values in synthetic Landsat values over a growing season (Figure 3)
revealed that phenological patterns were well captured; however, when seasonal differences lead to a change in land cover (i.e., disturbance, snow cover), the algorithm used to generate the synthetic Landsat images was, as expected, less effective at predicting reflectance.

**Paper in Review**

![Figure 1: Comparison of fine scale structure between observed Landsat scene (acquired July 09 2008) and predicted Landsat scene (predicted using July 12 as prediction date).](image)
Figure 2: Comparison between observed Landsat scene (left column), observed MODIS scene (central column) and predicted Landsat scene (right column) for the 5 acquisition dates. No synthetic data has been predicted for August 13 as the Landsat input data from August have been used as T1 image (Band combination: NIR, Red, Green).
Figure 3: Eight-day time series of NDVI values averaged for the most common land cover types found in the study area for DOY 120 to 280 (2001). The triangles represent synthetic NDVI predictions generated using STARFM. The open squares show NDVI observations generated from the real Landsat images at the four control dates. The bars represent the standard deviations from the mean NDVI values for observations and predictions, respectively. The grey line at DOY 226 marks the T1 date which was used for the STARFM predictions. Synthetic NDVI observations fit well into the annual pattern of the vegetation cycle for the different land cover types. Greatest differences in reflectance due to seasonal effects were found for the broadleaf vegetation class ($\Delta\text{NDVI}_\text{max}=0.21$), while the coniferous land cover type maintained a more homogeneous level of NDVI ($\Delta\text{NDVI}_\text{max}=0.16$).
1-3) Detecting And Validation Of Change Using Synthetic Landsat 25m 16 Day Time Series

Introduction
Natural and anthropogenic disturbances play a key role in habitat modeling and ecosystem functioning (Schimel et al. 1997; Hansen et al. 2001; Foster et al. 2003), and influence productivity and resource availability across a broad range of spatial and temporal dimensions. In forested environments, disturbance agents such as fire, insects, and various human activities related to settlement, cultivation, and resource extraction create pulses of biomass loss that influence biogeochemical cycling (DeFries et al. 1999; Patenaude et al. 2005; Morehouse et al. 2008, Masek and Collatz 2006), and exert a strong imprint on both habitat structure (Mladenoff et al. 1993; Spies et al. 1994; Turner et al. 1997) and the distribution of wildlife species (Foster et al. 2003; Nielsen et al. 2004; Linke et al. 2005, Lada et al. 2008; Leonard et al. 2008). As a result, detailed information on forest disturbance is important for a wide range of applications from ecological modeling to estimating carbon budgets on regional and global scales. Remote sensing is a critical data source for observing and understanding the effects of landscape disturbance (e.g. Potter et al. 2003; Linke et al. 2008a, Masek et al. 2008), but trade-offs in sensor designs that balance spatial detail with concerns for swath width and repeat coverage (Price 1994) can limit our capacity monitor changes effectively (e.g. Gao et al. 2006; Pape and Franklin 2008). Landsat, with a spatial resolution of 30 m and spatial extent of 185 x 185 km per scene, is used widely for mapping biophysical vegetation parameters (Cohen and Goward 2004; Masek et al. 2006) and has proven useful for monitoring land cover (Wulder et al. 2008; Linke et al. 2009) and ecosystem disturbance (Healey et al. 2005; Masek et al. 2006; Masek et al. 2008). The minimum 16-day revisit cycle of the platform, however, which can be markedly extended due to cloud contamination or duty cycle limitations (Ju and Roy, 2008) can create difficulties in capturing disturbance events in a timely manner (Gao et al. 2006; Leckie 1990; Pape and Franklin 2008). One technique for increasing the temporal frequency of high-spatial-resolution satellite observations is the blending of data from sensors with complementary spatial and temporal characteristics, with the aim of generating synthetic observations with both high spatial and temporal resolutions. The development of data fusion techniques has helped to improve the temporal resolution of fine spatial resolution data by blending observations from sensors with differing spatial and temporal characteristics. However, most data-fusion approaches are often based on spatially integrating reflectance observations and, as a result, are not specifically designed for mapping disturbance events, particularly if they occur in the sub-pixel range of the coarse-spatial-resolution image data.

This study introduces a new data fusion model for producing synthetic imagery and the detection of changes termed Spatial Temporal Adaptive Algorithm for mapping Reflectance Change (STAARCH). The algorithm is designed to detect changes in reflectance, denoting disturbance, using Tasseled Cap transformations of both Landsat TM/ETM and MODIS reflectance data. The algorithm requires a minimum of two Landsat scenes; one at the beginning and one at the end of the observation period, enabling the development of a change mask for input to the algorithm.
Delineation of disturbance features
The spatial delineation of disturbance features were determined by STAARCH at 30 m spatial resolution using a change mask derived from Landsat imagery obtained at the beginning and at the end of an observation period (Figure 4a,b). Change detection is performed using the Disturbance Index (DI) described in Healey et al. (2005), an index specifically designed to detect changes in forested land cover types (Figure 4c). The change mask (Figure 4c) is used to delineate the spatial extent of disturbance events occurring within the time frame represented by the Landsat image pair. The date of disturbance is then determined from a series of MODIS images, usually acquired at eight-day time steps between the dates of the Landsat images (Figure 5). STAARCH also uses a Landsat-derived land cover classification product to identify the expected land cover type of each pixel (from before the disturbance) and assess its reflectance relative to the average reflectance of the given land-cover class. Change detection in STAARCH is restricted to the vegetated land surface, with change features suppressed for non-vegetated land cover classes.
Figure 4 Comparison between the disturbance index (DI) computed for Landsat TM in 2002 (Figure 4a) and 2005 (Figure 4b). Dark DI values correspond to undisturbed areas, disturbances are highlighted in white. Figure 4c Landsat derived change mask between 2002/08/08 and 2005/09/21. The mask was derived from the difference in disturbance of the two Landsat images.
Figure 5: Sequences of eight-day disturbance intervals for the year of 2003 (Figure 5a), 2004 (Figure 5b) and 2005 (Figure 5c), respectively. The colors of the disturbed areas corresponds to the 8-day time intervals at which disturbances most likely occurred (derived from STAARCH).
Figure 6: Comparison between yearly disturbances derived from the validation data set and STAARCH determined year of disturbance. The date of disturbance for all polygons, that show the same color for outline and fill, has been correctly identified by the algorithm; polygons that show a different outline color than filling have been misclassified by STAARCH. The area of correctly identified disturbances was 87%, 87% and 89% in 2002, 2003 and 2005, respectively.

1-4) Proof of STAARCH over prototype study area
The algorithm has been tested over a 185 x 185 km study area in west-central Alberta, Canada. We acquired the validation data set from a disturbance inventory feature database developed by the multi-annual disturbance mapping project of Linke et al. (2009). This project delineated forest-replacing disturbance features in polygon-vector format from annual difference layers, derived from an annual series of summer Landsat Thematic Mapper and Enhanced Thematic Mapper Plus spanning the years 1998 to 2005. Results show that STAARCH was able to identify spatial and temporal changes in the landscape with a high level of detail. The spatial accuracy of the disturbed area was 93% when compared to the validation data set, while temporal changes in the landscape were correctly estimated for 87% to 89% of instances for the total disturbed area. The change sequence derived from STAARCH was also used to produce synthetic Landsat images for the study period for each available date of MODIS imagery. Comparison to existing Landsat observations showed that the change sequence derived from STAARCH helped to improve the prediction results when compared to previously published data fusion techniques (Figure 7).
Figure 7 A: Landsat scene observed at 2004/08/13. Figure 7b: STAARCH prediction based in the 2002 image using a spatial change mask derived from Landsat and a temporal detection of change features derived from MODIS. The STAARCH derived image (Figure 7b) includes the vast majority disturbance induced changes in the Landscape.

Accepted Paper
1-5) Attribution of Change

Introduction
As anthropogenic, landscape disturbances continue to increase in frequency and severity, there is a need for information to be more readily available for the forest manager. Remotely sensed imagery allows for disturbances to be detected at high spatial and temporal frequencies; however, the ability to label these disturbances is not as easy, often requiring manual interpretation and field visits. For many species, detecting the type of disturbance is equally important to management decisions, as detecting the disturbance itself. Grizzly bears, for example, have been known to utilize the edges of clear-cuts for foraging, but have shown markedly different reactions to numerous other disturbances, including roads, resource extraction well-sites and pipelines; all of which are common landscape disturbances.

This project attempts to develop a framework for automatically detecting disturbances, through a combination of high and medium resolution imagery, and labelling disturbances based on a combination of disturbance shape and reflectance attributes.

Detection of Changes
Following previous studies, differences were detected using two dates of the wetness band of a Tasselled Cap Transformation (TCT). Satellite images were converted to at satellite reflectance, normalized and TCT was calculated. Band subtraction provided a visualization of changes, allowing for extraction of disturbances. In order to eliminate noise generated in the change detection process, two thresholds were used to detect disturbances. The first threshold was set at four stand deviations above the mean TCT difference value; this extracts the brightest, most obvious changes. For the second threshold, all pixels above two and a half standard deviations above the mean are extracted, and these are calculated as second level changes, see Figure 8 for a comparison.

Figure 8 - Enhanced difference wetness index threshold comparison. Left image shows EWID image, white representing areas of high difference, or change. The middle image shows, in white, differences above 4 standard deviations, with above 3 standard deviations on the right.
Second level changes include a significant amount of noise. We applied edge detection algorithms on high resolution imagery in order to differentiate between real, and background changes. The premise of this comparison is that most apparent noise is often either linear features that are not being extracted, or false differences found in the centre of a homogenous region. If the detected changes are found near a detected edge, then we can assume that they are part of a linear feature. Using 5m resolution IRS imagery, edges are extracted and compared to second level disturbances from the wetness differencing. If disturbances are found to intersect with edges found in IRS imagery, they are accepted as valid changes, and converted to disturbance features, otherwise, the edges are discarded.

**Labeling of disturbances**

A model for labeling disturbances was developed based on a disturbance dataset generated by Linke et al. 2009. This dataset had all disturbances between the T1 and T2 satellite images manually identified and labeled through a combination of changed detection algorithms, manual elimination of noise, and digitization of features. The classification model was based on both reflectance attributes, derived from the original satellite image and the tasseled cap transformation, and shape based attributes. The incorporation of shape attributes allows for features with similar reflectance values to be distinguished, based on either their actual shape, or their relationship to other landscape features, such as distance to nearest roads, or distance to nearest \( k \) disturbance features. An example of a shape attribute is given in Figure 9.

![Example of elongation index](image)

**Figure 9** - Elongation index describing a shape attribute for disturbance features. The elongation index compares the overlap of the shape of the disturbance to a circle of equal area.

**Results**

Currently, the extraction of features using IRS edges is still underway. Multiple edge extraction methods have been attempted (spatial clustering, wombling, and wavelet) with varying levels of success. A model has been successfully developed, see Figure 10, and shows 95%-98% accuracy using cross-validation. This is significantly higher than models based on reflectance alone, illustrating the additional information derived from shape attributes.
Figure 10 - Classification model based on labeled, manually processed disturbances. At each step, disturbances are split based on a single variable threshold.
1-6) Develop models to predict canopy closure using synthetic Landsat for habitat modeling

Determination of canopy closure and leaf area is critical for a large number of modeling applications and characterization of wildlife habitats; leaf area is the primary interface of vegetation gas exchange with the atmosphere (Curran, 1992, White et al., 1997). Leaf area index (LAI) is defined as the amount of leaf surface present in a vegetation canopy over a given unit of ground surface area, usually measured in m² m⁻² (Black and Chen 1991). Canopy closure is also important for understanding biogeochemical cycling of important elements such as carbon and nitrogen in ecosystems because of litter drop and leaf retention, which are based on relationships between the leaf surface and its mass (Reich et al. 1995, White et al., 1997).

Remote sensing observations can be used to obtain estimates of leaf area from the proportion of absorbed radiation, which has been demonstrated to be closely related to the density of the absorption surface, i.e. the crown canopy (Sellers 1995, Tucker et al., 1979, Curran, 1992; Fassnacht, 1997; Stenberg, 2004). White et al. (1997) demonstrated the use of Landsat-5 TM imagery acquired over a study area in Glacier National Park (Montana, USA) for prediction LAI from Landsat derived vegetation indices. So far, canopy closure has been predicted for the study area on an annual basis using algorithms developed at the University of Calgary. In this study, the STAARCH algorithm described earlier in this report will be used to generate high temporal resolution synthetic reflectance estimates of the study area. This high density time series of synthetic-Landsat imagery will serve as a basis for determining canopy closure over the study site at 8-day time intervals. We will build upon existing algorithms to extract LAI and canopy closure from the generated STAARCH imagery.

White et al (1997) used two standard vegetation indices, the normalized difference vegetation index (NDVI) and the simple ratio (SR), computed from TM bands 3 (b3; 630-690 nm) and 4 (b4; 760-900 nm), where:

\[
\text{NDVI} = \frac{(b4 - b3)}{(b4 + b3)} \quad \text{and} \quad \text{SR} = \frac{b4}{b3}.
\]

Reflectance in SR and NDVI will, besides the canopy density, also depend on the canopy clumping, describing the deviation of the actual leaf distribution from a random distribution. Foliage clumping is dependant on species and age class and can be obtained from regressing estimates of stand DBH with sapwood (White et al., 1997). In this study we will build upon previously published estimates of canopy clumping for the main species and age classes observed in the study area. Forested areas are being classified into coniferous, broadleaf and mixed forests, following the EOSD land cover classification product. For each land cover class, a different regression equation will be applied, based on NDVI and/or SR. The results of these analyses will be a series of 8-day LAI images for the entire growing season. This will allow us to compare between and within forest classes the seasonal changes in LAI.
1-7) Role of understory as a driver of conifer canopy spectral reflectance, influence on capture of phenology `Phenological Cameras’

Phenological camera status
To date five cameras have been acquired; two purchased by CFS, three purchased by UBC and one purchased by UofC. Temperature data loggers will also be acquired for each – Scott Nielsen proposed the DS1921G model. The cameras are being camouflaged to make them less noticeable when deployed.

Sampling design
Locations along a transect near Hinton, Alberta, have been chosen based on elevation and vegetation communities. Two cameras will be deployed at each of three different elevations (e.g., ~1,000, ~1,350, and ~1,700 MASL). For each elevation, one camera will be located at a site with a high degree of expected phenological change, and the second in an area with a low degree of change (Table 1; Figures 11-16).

Table 1. Summary table describing six possible sites. More potential sites will be identified prior to departing for the field

<table>
<thead>
<tr>
<th>Camera No.</th>
<th>Photo-interpreted forest type</th>
<th>EOSD land cover class</th>
<th>Natural Sub-region</th>
<th>Elevation (metres above sea level)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Coniferous</td>
<td>Coniferous dense</td>
<td>Lower foothills</td>
<td>1,005</td>
</tr>
<tr>
<td>2</td>
<td>Mixed</td>
<td>Broadleaf dense</td>
<td>Lower foothills</td>
<td>987</td>
</tr>
<tr>
<td>3</td>
<td>Coniferous</td>
<td>Coniferous dense</td>
<td>Upper foothills</td>
<td>1,532</td>
</tr>
<tr>
<td>4</td>
<td>Mixed</td>
<td>Broadleaf dense</td>
<td>Upper foothills</td>
<td>1,522</td>
</tr>
<tr>
<td>5</td>
<td>Coniferous</td>
<td>Coniferous dense</td>
<td>Subalpine</td>
<td>1,783</td>
</tr>
<tr>
<td>6</td>
<td>Mixed</td>
<td>Broadleaf dense</td>
<td>Subalpine</td>
<td>1,917</td>
</tr>
</tbody>
</table>

Although Table 1 shows only six sites, several (e.g. 5-6) will be chosen for each natural sub-region which can be used as alternates. For instance, locating mixed forest in the subalpine region is particularly difficult, and having several potential sites from which to choose should make field work more efficient. Scott Nielsen has expressed interest in accompanying the installation crew, and his knowledge of the area will be critical for final site selection.
Figure 11. Possible locations for plots selected based on elevation and vegetation type.
Figure 12. Examples of possible plots selected based on elevation and vegetation type. Refer to Table 1 for plot description, and Figure 11 for locations. Aerial imagery has a spatial resolution of 0.20 m.
Figure 13. One of the wildlife tree retention block shown in Figure 12, image 2.

Figure 14. Typical mixed forest stand found in the lower foothills
Figure 15. The pine stand shown in Figure 12. This stand is representative of a large proportion of the vegetation found along the transect within the upper foothills and subalpine subregions.
1-8) Edges and transitions: Development of landscape edge features from continuous variables

Introduction
Edges between differing land cover or functional vegetation classes represent important transition zones for abiotic and biotic communities. Due to the impact of edges on many ecological processes, knowledge of their location and severity becomes important in measuring impacts of anthropogenic disturbances. There are many methods for determining the locations of edges in modern ecology, including spatial clustering, split moving window, wombling, and wavelet. While many of these methods have their own merits, this project focuses on understanding both the location and strength of edges in the landscape. As such, we will implement wombling as our edge detection method, and compare the edges detected in this method to edges extracted from a spatially clustered dataset.

Wombling
Wombling was first proposed by Womble (1951) as a method of measuring rates of change across a regular lattice. Womble proposed a quantitative measure of change by calculating the absolute value of the first derivative of the image. A 2 x 2 window is moved across a lattice dataset, and a function describing the plane of that window is created, see equation 1. The first partial derivative of the plane is given in equation 2, this is the slope of that plane, and is considered the rate of change for the 2 x 2 window. The angle of that rate of change is given in equation 3. The coordinates of the window are scaled to run from 0 to 1 in both the x and y directions, and the value of m and θ are calculated at the centroid of the window, where x and y = 0.5.

\[ f(x,y) = A(1-x)(1-y) + B(x)(1-y) + C(x)(y) + D(1-x)(y) \]  
\[ m = \sqrt{\left(\frac{\partial f(x,y)}{\partial x}\right)^2 + \left(\frac{\partial f(x,y)}{\partial y}\right)^2} \]  
\[ \theta = \arctan\left(\frac{\partial f(x,y)}{\partial x}/\frac{\partial f(x,y)}{\partial y}\right) + \Delta \]  
\[ \Delta = 0^\circ \quad \text{if} \quad \frac{\partial f(x,y)}{\partial x}/\frac{\partial f(x,y)}{\partial y} > 0 \]  
\[ = 180^\circ \quad \text{if} \quad \frac{\partial f(x,y)}{\partial x}/\frac{\partial f(x,y)}{\partial y} < 0 \]

In order to classify cells as edge or not edge, an extraction algorithm was applied; cells defined as edges were classed as a Boundary Element (BE), and were given a categorical strength (high, medium, medium-low and low). Wombling values were classified into deciles, and the top four classes were extracted for analysis. A 3 x 3 window was passed over the dataset, for each cell, if a surrounding cell was in the same decile or higher, and had an angle of ± 30°, then it is classified as a BE. Based on its wombling strength, it is given a strength class.
Comparison of Wombling to EOSD

One purpose of this research is to investigate whether different edge types display different severities of edge strength. In order to evaluate this, transition classes were extracted from the Earth observation for sustainable development of forests (EOSD) dataset. The EOSD dataset was re-sampled to 30 m pixels (from 25 m), in order to match the wombling dataset (derived from Landsat data), and every cell was given a transition code, based on the Level 4 EOSD classification, see Figure 16.

For each EOSD transition class, mean, coefficient of variation and frequency of High and Medium strength boundary elements were calculated, based on the underlying wombling values. A plot of these values (Figure 16) shows that, indeed, there is dramatic variation between class strength. The colours in the plot indicate the transition type when the edges are classified as forest, non-forest and other. This indicates that edges extracted from the EOSD can show dramatic variation in their severity, even within groups that are normally extracted as one unique class type.

Wombling was also compared to the EOSD product by comparing Landscape Pattern Indices (LPIs) calculated on a forest / non-forest layer derived from the EOSD product. LPIs are common methods of measuring fragmentation across a landscape, here, we are attempting to show that wombling can describe many of the same disturbances identified through LPIs, but also provide important information about local disturbances. See Figure 17 for a comparison of LPIs calculated at approximately 1km scale, compared to the same area showing wombling values.
Figure 17 - Comparison of LPI and wombling values for the same area at 1:250,000. White indicates high edge density in the LPI image on the left, and indicates strong wombling values on the right.

Results
As described above, transition classes defined through the EOSD show dramatic variation, indicating that they may not represent the same ecological edge as other edges in the same class. This is an important realization in ecological modeling; edge type is insufficient in model effects of edges in a landscape. The possible applications of combining both edge type and strength are wide reaching, and could have serious impacts in many ecological models.
1-9) Landsat Alternatives: Cross-sensor STAARCH

Although the STAARCH algorithm uses Landsat data as an input, recent technical problems with the two existing satellites and delays in the launch of their successor have motivated the search for alternative data sources for wide area mapping. Two potential platforms that merit investigation include the Advanced Wide Field Sensor (AWiFS) and the Satellite Pour l’Observation de la Terre (SPOT), the characteristics of which are summarized in Table 2. Single AWiFS and SPOT scenes were acquired over the area of interest in the summer of 2008 and will be tested as inputs for the STAARCH algorithm in combination with Landsat data (Figure 18).

Table 2. Comparison between AWiFS, SPOT, and Landsat satellites.

<table>
<thead>
<tr>
<th>Sensor</th>
<th>Swath Width (km) or Scene Dimensions (km x km)</th>
<th>Return Period</th>
<th>Spatial Resolution</th>
<th>Band Wavelengths (μm)</th>
<th>Radiometric Resolution</th>
</tr>
</thead>
<tbody>
<tr>
<td>IRS ResourceSat-1 AWiFS</td>
<td>740</td>
<td>~ 2 weeks</td>
<td>56 m multispectral</td>
<td>0.52-0.69, 0.62-0.68, 0.77-0.86, 1.55-1.70</td>
<td>10 bit</td>
</tr>
<tr>
<td>SPOT-2 HRV</td>
<td>60 x 60</td>
<td>26 days; 1 and 4 days alternatively when viewing off-nadir</td>
<td>20 m multispectral</td>
<td>0.60-0.59, 0.61-0.68, 0.78-0.89, 0.50-0.73 (pan)*</td>
<td>8 bit</td>
</tr>
<tr>
<td>SPOT-4 HRVIR</td>
<td>10 m panchromatic</td>
<td>16 days</td>
<td>10 m panchromatic</td>
<td>0.60-0.59, 0.61-0.68, 0.78-0.89, 1.58-1.75, 0.61-0.68 (pan)</td>
<td></td>
</tr>
<tr>
<td>Landsat 5 TM</td>
<td>185</td>
<td>16 days</td>
<td>30 m multispectral, 120 m thermal</td>
<td>0.45-0.52, 0.62-0.60, 0.63-0.69, 0.76-0.90, 1.55-1.75, 10.4-12.5, 2.08-2.35</td>
<td>8 bit</td>
</tr>
<tr>
<td>Landsat 7 ETM+</td>
<td>60 m thermal, 15 m panchromatic</td>
<td></td>
<td></td>
<td>0.450-0.515, 0.525-0.605, 0.630-0.690, 0.750-0.900, 1.55-1.75, 10.40-12.50, 2.09-2.35, 0.50-0.90 (pan)</td>
<td></td>
</tr>
</tbody>
</table>
Figure 18  Maps displaying SPOT 4 and AWiFS images acquired during the summer of 2008.
2-1) Spectral trajectories approach upon synthetic Landsat data predict likelihood of RA within year

The current outbreak of mountain pine beetle (*Dendroctonus ponderosae*) in western Canada has been increasing over the past decade and is currently estimated to be impacting over 13 million hectares of forest with varying levels of severity. Large area insect monitoring is typically undertaken using manual aerial overview sketch mapping, whereby an interpreter depicts areas of homogenous insect attack conditions onto 1:250,000 or 1:100,000 scale maps. These surveys provide valuable strategic data for management at the provincial scale. The coarse spatial and attribute resolution of these data however, make them inappropriate for fine-scale monitoring and operational planning. For instance, it is not possible to estimate the initial timing of attack and year of stand death. In this study, we utilize eight Landsat scenes collected over a 14 year period in north-central British Columbia, Canada, where the infestation has gradually developed both spatially and temporally. After pre-processing and normalizing the eight scenes using a relative normalization procedure, decision free analysis was applied to classify spectral trajectories of the Normalized Difference Moisture Index (NDMI) (Figure 19). From the classified temporal sequence of images, key parameters were extracted including the presence of beetle disturbance and timing of stand decline. The accuracy of discriminating beetle attack from healthy forest stands was assessed both spatially and temporally using three years of aerial survey data (1996, 2003, and 2004) with results indicating overall classification accuracies varying between 71 and 86%. As expected, the earliest and least severe attack year (1996), recorded the lowest overall accuracy. The relationship between the timing of stand attack (i.e. moderate to severe beetle infestation) and NDMI (initial year of detected disturbance) was also explored. The results suggest that there is potential for deriving regional estimates of the year of stand death using Landsat data and decision tree analysis however, a higher temporal frequency of images is required to quantify the timing of mountain pine beetle attack.
Figure 19: Image subset of Morice TSA displaying MPB attack between 1993 and 2006. Note: the RGB band combination used was Landsat TM bands 5, 4, and 3 with a 2 SD stretch. Dark green areas represent healthy forest, reddish areas MPB, and harvested areas light green or magenta.
Published Paper:

2-2) Infestation maps over MPB area: Assessment of impacts of MPB and GB home ranges in the Alberta foothills.

Introduction
Numerous studies have shown that moderate spatial resolution remote sensing imagery can be useful for change detection and characterizing forest disturbance (Coppin and Bauer 1994, Adams et al. 1995, Fraser and Li 2002, Kennedy et al. 2007). Recently, Landsat-5 Thematic Mapper (TM) and Landsat-7 Enhanced Thematic Mapper Plus (ETM+) imagery have been proven effective in mapping the mountain pine beetle infestation in British Columbia, and also demonstrating an applicability elsewhere in western North America (Skakun et al. 2003, Coops et al. 2006, Wulder et al. 2006). Typically based on indices derived from Tasseled cap transformations (TCT), these studies have been successful in producing both discrete (categorical) and probabilistic infestation maps.

The following report describes the methods and results of the 2007 red attack classification map created in support of the Foothills Model Forest Grizzly Bear Research Program (FMFGBRP).

Data
Airborne digital image data
Airborne digital imagery were acquired across two transects during the summer of 2008. The images are true colour with a spatial resolution, or pixel size, of 0.20 m. The two transects totaled 200 km in length, with a swath width of approximately 750 m. The airborne images were used chiefly to support algorithm calibration efforts through identification of healthy forest stands. Figure 20 provides examples of the imagery and displays the typical intensity of the mountain pine beetle infestation across much of the study area. Using the aerial imagery, 497 locations exhibiting no visible signs of red attack were selected for training and validation purposes. Trees healthy in 2008 can be inferred to have also been un-infested in 2007, thus suitable for calibration of year 2007 conditions.

Helicopter global positioning system survey data
Tactical infestation surveys are undertaken by experienced surveyors from helicopters whereby the location (using global position systems, GPS) and the number of trees at a given site are recorded. These Helicopter-based surveys, or Heli-GPS, provided infestation data depicting mountain pine beetle red attack across the area of interest. The survey data, collected in 2007, consists of discrete points with the number of red crowns at a given location stored as an attribute, and were used to identify infested areas. Figure 21 shows the number of red attack crowns found at each survey point. While infestation was widespread across the area of interest, intensity at any given location was relatively light, with most locations having fewer than 10 crowns exhibiting signs of red attack (Figure 1, Figure 2).
Figure 20. Examples of the 0.20 m spatial resolution airborne images collected along two transects. Infestation intensities shown here are typical of the area of interest.

Landsat Imagery
Landsat images were acquired for both 2006 and 2008. Time 1 (T1) data consisted of two scenes acquired on 2 September 2006, and Time 2 (T2) data consisted of two scenes collected on 6 August 2008. Both scene pairs were acquired from path 46, rows 22 and 23 (Figure 22). The Landsat-5 TM scenes were processed following the methods outlined in Wulder et al. (2005). First, the scene pairs were mosaicked into single date images. Shadow, snow and other non-forest areas were masked out using band thresholding and classification and regression tree analysis, while cloud and haze were manually delineated (Figure 23).
Figure 21. Number of red attack crowns found at locations mapped during the 2007 Heli-GPS survey. Note that for display purposes, the Y-axis has been limited to 2,000 observations, while the number of observations in the 1-10 crown category is 33,677.

Figure 22. Time 1 and T2 Landsat data used to create the mountain pine beetle red attack map for 2007. Note the large amount of cloud in the T1 image, and the location of an inset map.
The images were converted to top of atmosphere (TOA) reflectance values using published post-launch gains and offsets (ITT 2008). The T1 mosaic was georeferenced to a National Topographic System roads map using a nearest neighbour algorithm and a second-order polynomial transformation (26 ground control points, RMSE = 0.86 pixels or 26 m). The T1 and T2 mosaics were then co-registered using a nearest neighbour, second-order polynomial transformation (29 ground control points, RMSE = 0.48 pixels or 14 m). The T1 and T2 mosaics were then co-registered using a nearest neighbour, second-order polynomial transformation (29 ground control points, rmse = 0.48 pixels). The T2 image was normalized to T1 using mean TOA reflectance values collected from bright and dark pseudo-invariant targets from each image. Wetness indices were calculated using a Tasseled cap transformation (TCT); the coefficients applied to the individual bands are shown in Table 3. An enhanced wetness difference index (EWDI) was then calculated by subtracting T2 wetness from T1 wetness.

Figure 23. Example of the forest/non-forest mask employed during the analysis. Non-forest areas are shaded black in the right hand panel.
Table 3. Landsat-5 TM Tasseled cap transformation coefficients.

<table>
<thead>
<tr>
<th>Feature</th>
<th>Band 1</th>
<th>Band 2</th>
<th>Band 3</th>
<th>Band 4</th>
<th>Band 5</th>
<th>Band 7</th>
</tr>
</thead>
<tbody>
<tr>
<td>Wetness</td>
<td>0.151</td>
<td>0.197</td>
<td>0.328</td>
<td>0.341</td>
<td>-0.711</td>
<td>-0.457</td>
</tr>
</tbody>
</table>

Enhanced wetness difference index analysis
To characterize healthy forest, suitable sites were identified using the high spatial resolution aerial imagery (n = 343), and their corresponding EWDI values were then extracted. Validation data were collected in the same fashion (n = 154). To distinguish infested stands, EWDI values were extracted from sites where the Heli-GPS data indicated infestation was relatively severe (n= 68). Validation data were collected by selecting Heli-GPS points with high red crown counts (n = 154). Following procedures described in Wulder et al. (2005) thresholds were then iteratively applied to the EWDI and then validated to derive a map of areas likely experiencing red attack.

Results and Discussion
Figure 24 displays the EWDI distributions of healthy and infested stands. Testing various EWDI thresholds for the EWDI indicated that values between 0.02-0.03 were characteristic of infested locations. The resulting classification has an overall accuracy of 68% (Table 4). More importantly, the producer’s accuracy for red attack is 47%, indicating that over half the reference red attack locations were incorrectly assigned to the healthy class. The user’s accuracy, however, indicates that 82% of the sites classified as red attack were correct, while 18% were incorrectly included in the non-attack class. Figure 25 provides a comparison between the Heli-GPS survey data and the final red attack classification for a portion of the study area.

Within the areas classified as forest, which is approximately 2,100,000 ha, the Heli-GPS data identifies 6,430 ha within which red crowns are present, while the classification identifies 13,670 ha. It is unclear if the discrepancy is the result of an overestimation of infestation amounts by the classification, or if beetle damage is being detected in areas not mapped by the Heli-GPS survey.

While the capacity to map mountain pine beetle infestation using moderate resolution remote sensing data is well-established, success depends on the timing and severity of attack. For instance, Skakun et al. (2003) found significant accuracy improvements for sites containing 30-50 read attack trees versus those containing 10-29. While widespread, the low severity of the infestation is a confounding factor. Nonetheless, the low cost large area coverage of Landsat data, and the proven capability of Landsat-derived products, makes it a critical tool for forest health monitoring. Further, the utility of these maps is increased when combined with survey information from other sources.
and forest inventory based information on susceptibility to mountain pine beetle infestation.

Figure 24. Box and whisker plot displaying mean, standard error of the mean, and ±two standard deviations of the enhanced wetness difference index for selected healthy and attacked sites.

Table 4. Confusion matrix summarizing the accuracy of the red attack classification.

<table>
<thead>
<tr>
<th>Model</th>
<th>Reference</th>
<th>Not red</th>
<th>Red</th>
<th>Row total</th>
<th>Producer's accuracy</th>
<th>Omission</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Not red</td>
<td>138</td>
<td>16</td>
<td>154</td>
<td>89.61%</td>
<td>10.39%</td>
</tr>
<tr>
<td></td>
<td>Red</td>
<td>82</td>
<td>72</td>
<td>154</td>
<td>46.75%</td>
<td>53.25%</td>
</tr>
<tr>
<td>Column total</td>
<td></td>
<td>220</td>
<td>88</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>User's accuracy</td>
<td></td>
<td>62.73%</td>
<td>81.82%</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Commission</td>
<td></td>
<td>37.27%</td>
<td>18.18%</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Overall Accuracy 68.18%

<p>| | |</p>
<table>
<thead>
<tr>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>True positive rate</td>
<td>46.75%</td>
</tr>
<tr>
<td>False positive rate</td>
<td>10.39%</td>
</tr>
<tr>
<td>True negative rate</td>
<td>89.61%</td>
</tr>
<tr>
<td>False negative rate</td>
<td>53.25%</td>
</tr>
<tr>
<td>Precision</td>
<td>81.82%</td>
</tr>
<tr>
<td>N</td>
<td>308</td>
</tr>
</tbody>
</table>
Figure 25. Comparison of the 2007 Heli-GPS survey data (right) and the final red attack classification (left).

A pilot study investigating the linkages between bear habitat usage and landscape disturbance related to mountain pine beetle attack and associated mitigation and harvesting activities is currently underway. Annual bear home range data is being employed to stratify the landscape in an overlay exercise combining maps of red attack, mitigation and forest disturbance to identify related patterns of habitat use (Figure 26).
Figure 26. Examples of home ranges used by two bears in 2007, and Landsat image exhibiting land cover and disturbance.
TOPIC 3: ONGOING MONITORING OF MPB INFESTATION AND COLLECTION OF GROUND VALIDATION DATA

3-1) Field program: Mensuration (Pine, red attack, DBH, Age) over prototype area; airborne data over full study area

Field data collection was undertaken to assist the University of Calgary project partners with a five year study designed to investigate the effects of the mountain pine beetle infestation and related mitigation activities on grizzly bear populations, and thereby produce additional data for model building and remote sensing related mapping.

A total of 250 plot locations had been selected through proportional stratified random sampling, and a subset of 25 of these sites was provided to us as a shapefile. Of these, 23 locations were visited and data was collected for 20 plots. Several provided plot locations had to be rejected, as recent harvesting activities had cleared extensive areas. Where possible, plots were moved into adjacent remaining stands, provided they were uniform and large enough to merit a plot.

Each of the 30 metre square plots was placed in a uniform and representative forest area as close to the provided location as possible, and a GPS location was taken for the plot centre. Data on the general plot characteristics, including soil moisture and land cover, were recorded and a representative horizontal photo was taken of the plot. At each of the four plot corners, as well as at the plot centre, vertical photos were taken with a hemispherical camera to provide a record of the canopy closure. A variable radius plot (using a BAF 2 or 4 prism, depending on stand density) was measured from the plot centre, and the species and diameter for each tree within the plot was recorded. Additionally, for two trees of each species, a measure of the height was taken using a vertex hypsometer, and an increment core was taken to provide the age of the tree.

Determination of field sites for phonological camera network

In addition to the field program described above, a reconnaissance of six potential sites for monitoring vegetation phenology using time-lapse photography was undertaken during August of 2008. The sites are located along a 90 km transect in western Alberta near the town of Hinton and immediately east of Jasper National Park (Figure 27). The sites range in elevation from approximately 1,000 to 2,000 m above sea level, and are located in the lower and upper foothills, and the sub-alpine natural subregions of Alberta. True colour digital aerial imagery with a spatial resolution of 0.20 m (Figure 28) and a swath width of approximately 600 m was acquired along the transect in August of 2008. This report includes general descriptions of the six sites, and recommendations on possible camera placement and scene capture. (Note that all of the photographs were taken using the same type of camera that will be set up at the sites.)
Figure 27. Perspective view of the phenological camera transect (green line) and the six proposed sites (red cylinders). View is towards the southwest, scale is variable, vertical exaggeration = 3x.

Figure 28. Examples of the 0.20 m spatial resolution airborne images collected along two transects. Infestation intensities shown here are typical of the area of interest.
Site descriptions
Table 5 provides a summary of the sites. Because some of the suggested sites were only approximate coordinates and the imagery was not available prior to the field program, two of the sites are not located within the image swath.

Table 5. Summary descriptions of the six phenological camera sites.

<table>
<thead>
<tr>
<th>Camera ID</th>
<th>Site name</th>
<th>Elevation (masl)</th>
<th>Natural subregion</th>
<th>Overstory species</th>
<th>Understory species and ground cover</th>
<th>Notes</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Fickle Lake</td>
<td>996</td>
<td>Lower foothills</td>
<td>Mixed trembling aspen and spruce</td>
<td>Poplar, aspen fireweed, grasses, other flowering species</td>
<td>Harvested area with wildlife tree retention patches. Opportunity to monitor overstory and regenerating deciduous species.</td>
</tr>
<tr>
<td>2</td>
<td>Bryan Sprur</td>
<td>1,112</td>
<td>Lower foothills</td>
<td>Mixed pine, spruce, aspen</td>
<td>Cranberry, trappers or Labrador tea, soopalalie, grasses, moss</td>
<td>Opportunity to monitor overstory and understory deciduous species.</td>
</tr>
<tr>
<td>3</td>
<td>Hwy 40</td>
<td>1,331</td>
<td>Upper foothills</td>
<td>Spruce, some cottonwood</td>
<td>Alder, willow, soopalalie, vetch, moss</td>
<td>Site not located within imagery, may be difficult to relocate.</td>
</tr>
<tr>
<td>4</td>
<td>Watson Creek</td>
<td>1,468</td>
<td>Upper foothills</td>
<td>Pine</td>
<td>Spruce, soopalalie, trappers or Labrador tea, grasses, moss</td>
<td>Pure pine site, but representative of surrounding forest.</td>
</tr>
<tr>
<td>5</td>
<td>Prospect Creek</td>
<td>1,706</td>
<td>Sub-alpine</td>
<td>Pine, with some spruce, cottonwood, aspen, and willow in midstory</td>
<td>Grasses, lichen, some moss</td>
<td>Site not located within imagery, but easy to relocate. Not accessible by automobile until after snow melt.</td>
</tr>
<tr>
<td>6</td>
<td>Cardinal Divide</td>
<td>2,026</td>
<td>Sub-alpine</td>
<td>Spruce</td>
<td>Willow, scrub birch, grass, lichen, moss, some exposed gravel</td>
<td>Cannot mount camera on tree. Not accessible by automobile until after snow melt.</td>
</tr>
</tbody>
</table>

Fickle Lake
The Fickle Lake site is located in a recently harvested area containing scattered wildlife tree retention patches (Figure 29). The wildlife tree patches consist of mixed spruce and trembling aspen, while the harvested areas contain one or two year old regenerating spruce, aspen, and poplar, with fireweed, grass, and other flowering plants also present. This site is unique because it presents an opportunity to monitor phenological changes in both overstory deciduous species in the wildlife tree patches, and the regenerating ground cover in the cut areas (Figure 30).

---

1 All elevations were measured using an autonomous handheld GPS and may only be accurate to within 30-60 vertical metres.
Figure 29. Aerial image of the Fickle Lake site, which consists of a cut block with numerous wildlife tree retention patches.
Figure 30. The Fickle Lake site is located in a harvested area containing one to two year old regenerating vegetation and numerous wildlife tree patches; images similar to these could be used to simultaneously monitor phenology in both communities.
Bryan Spur
The Bryan Spur site is located in a mixed stand north of the town of Robb (Figure 31). Overstory species include aspen, pine, and spruce. Ground cover consists of cranberry, soopolallie, Labrador tea, moss, and leaf litter. Numerous game trails cross through the site, and several scraped trees indicate that bears are present in the area. A camera could be mounted high on a tree stem to capture phenological change in both the overstory and understory (Figure 32), but care should to be taken to locate it away from a game trail to ensure security.

Figure 31 Aerial image of the Bryan Spur site. A camera could be placed near the centre of the map
Figure 32. The Bryan Spur site is located in a mixed species stand; images similar to these could be used to simultaneously monitor phenology in both the overstory and understory.
**Hwy 40**
The site chosen in the field was located approximately 400 m outside of the aerial imagery swath. The site is dominated by spruce, though some cottonwood is present in the overstory. The understory consists of alder and willow shrub, and the ground cover of soopolallie, vetch, grass and moss (Figure 33). A decision will have to be made whether to (1) use the selected site, (2) find a new location within the image swath, or (3) discard this site entirely. The HWY 40 site was recommended by Scott Nielsen because it brackets the Bryan Spur and Watson Creek sites, as there is a large elevation change between the two. The area seems to be extensively used by campers and all terrain vehicle riders, so it may be difficult to find a secure location for camera placement.

![HWY 40 site's overstory consists of spruce with some isolated cottonwood (top). The understory includes alder and willow up to three metres tall.](image)

**Watson Creek**
The Watson Creek site is located within a pure pine stand immediately south of Highway 40 (Figure 34). Ground cover includes soopolallie, Labrador tea, grass, and moss (Figure 35). Immediately south of the site, a gully containing willow shrub trends southwest to
northeast. Although no deciduous species are found in the overstory, the site seems to be very representative of the surrounding landscape; specifically, pure pine stands with low understory vegetation. A discussion on its value as a monitoring site should be held, as while it contains little in the way of deciduous overstory vegetation, it is typical of the surrounding forests.

Figure 34. Aerial image of the Watson Creek site. A camera could be placed near the centre of the map.
Figure 35. The Watson Creek site is located in a pure pine stand (top). With little to be gained from monitoring the overstory, a camera could be placed to only capture ground vegetation (bottom).
Prospect Creek
The Prospect Creek site is the second not located within the image swath. Unlike the HWY 40 site, however, the site is only 30 m outside the swath and can be easily moved (Figure 36). The site is located on a south facing hillside with a slope of approximately 40 degrees, and is immediately north of Prospect Creek. Although dominated by pine, some spruce, cottonwood, and aspen are present in the midstory, while the ground cover is composed of grass, lichen, and moss (Figure 37). The Prospect Creek and Cardinal Divide sites are not accessible by automobile until after snow melt, which may not be until late May.

Figure 36 Aerial image of the Prospect Creek site. A camera could be placed near the centre of the map
Figure 37. The Prospect Creek site is located in an open stand dominated by pine, but containing some deciduous species in the midstory. Images for monitoring phenology would be similar to these.
Cardinal Divide
The Cardinal Divide site is located approximately 2,000 m above sea level near the transition between the sub-alpine and alpine natural sub-regions. The site consists of isolated islands of spruce trees surrounded by meadows of scrub birch, willow, pine krummholz, moss, lichen, and some exposed gravel (Figures 38 and 39). The site is perhaps the most unique of the six. The trees are not large enough to have a camera mounted on them, so an alternative solution will have to be found. A camera could be mounted on a fence pole; however, the openness of the area would make it easy to spot from a long distance. Like Prospect Creek, this site will not be accessible by automobile until late May.

Figure 38. Aerial image of the Cardinal Divide site. A camera could be placed within the stand of trees immediately south of the centre of the map.
Figure 39. The Cardinal Divide site is composed to isolated spruce trees surrounded by open meadow. The PhD candidate is 1.8 m tall (top).
3-2. Development of ongoing airborne program to collect MPB, growth, and species data

High-spatial resolution digital aerial images were acquired over two study sites, the Hinton Transect (Figure 40) and the Northern Transect (Figure 41), using a Canon EOS-1Ds Mark II camera, fitted with a bayer pattern filter, mounted on a fixed wing aircraft. The camera uses a complementary metal–oxide–semiconductor (CMOS) sensor which provides an effective resolution of 16.7 megapixels. Imagery was acquired near-nadir over the centre point of eight of the variable radius plots at a flying height of 1100 m, with a focal length of 85 mm, resulting in a 20 cm spatial resolution. The time for image acquisition was chosen to be close to solar noon to reduce the bidirectional reflectance effects in the imagery. Data were georectified using a QuickBird multispectral (2.44 m spatial resolution) image projected in UTM North American Datum 83. Image coordinates were supplied by an onboard GPS coupled with an inertial navigation system to assist accurate georectification. Each image covered an area of approximately 0.14 km² (0.44 km x 0.31 km or 4850 x 3110 pixels). Imagery was recorded in 3 channels representing the spectral ranges which approximate to: 0.4 – 0.5 μm (blue), 0.5 – 0.6 μm (green), and 0.6 – 0.7 μm (red). Table 6 shows an overview of the two transects flown for areal photography.

<table>
<thead>
<tr>
<th>Transect</th>
<th>Specifications</th>
</tr>
</thead>
<tbody>
<tr>
<td>Hinton transect</td>
<td>Resolution: 20cm, True color composites</td>
</tr>
<tr>
<td></td>
<td>Length: 90 km</td>
</tr>
<tr>
<td>Northern Transect</td>
<td>Resolution: 20cm, True color composites</td>
</tr>
<tr>
<td></td>
<td>Length: 118 km</td>
</tr>
</tbody>
</table>
Figure 40. Map of the Hinton transect. The transect is 90 km long, and sites range in elevation from approximately 1,000 to 2,000 m above mean sea level. The transect is following the sites of the phonological cameras described above.
Figure 41. Map of the Northern transect.
TOPIC 4: PROJECTING LANDSCAPE RECOVERY AND CHANGE

Given that significant disturbance is occurring on the landscape, due to harvesting, development and beetle activity, it is critical to understand how the landscape will recover through time. For instance, what is the future state of forest, the growth rates, size and condition of forest patches, understory development and availability, and the impact on future grizzly bear habitats and populations? In short, what is the forest landscape state in 2020, 2050 and 2080? To examine these issues, we will implement a scenario-based modeling approach which combines information on land cover, disturbance, patch dynamics, and changes in forest growth and recovery growth rates.

4-1: Review of options for combining remotely sensed data and spatial data layers to project landscape recovery and change

Climate data were extrapolated across the Pacific Northwest using the CLIMATE-BC methodology, which includes a modified PRISM (Parameter-elevation Regressions on Independent Slopes Model) algorithm to improve estimates of precipitation and temperature variation in mountainous terrain (Hamann and Wang, 2005). To derive monthly minimum and maximum temperate extremes and rainfall, a 90m Digital Elevation Model (DEM) was obtained from Shuttle Radar Topography Mission (SRTM) and resampled to 100m and to 1-km base for the analysis. The number of days per month with subfreezing temperatures was estimated from empirical equations with mean minimum temperature (Coops et al., 1998). Monthly global solar radiation was derived using a topographic solar radiation model based on a regionally defined cloudless index, which was used to downscale the North American Regional Reanalysis (NARR). This re-analysis consists of a three hourly radiation budget for North America from 1979 to the 2008. We averaged the three hourly observations into spatial coverages of monthly observations following methods of Schroeder et al., (2008).

Changes in climate were derived from the Special Report on Emission Scenarios (SRES) climate scenarios developed by the IPCC (IPCC, 2000). These scenarios include “a business as usual” prediction that includes a number of economic and ecological assumptions and a revised assessment of the impact of emissions. We applied, similar to other researchers (Monserud et al., 2008) the Canadian Climate Centre (CGCM2; Flato et al., 2000) predictions based on the A2 scenario which can be viewed as close to the upper bound of the SRES scenarios (IPCC 2000; Monserud et al., 2008). We used three decadal time periods, the 2020’s (2011-2040), 2050’s (2041 – 2070) and the 2080’s (2071 – 2100). The scenarios are provided in terms of anomalies from the baseline period and as a result they are integrated with the CLIMATE-BC layers described above. As average irradiance from CGCM2 during future base periods does not differ substantially from 20th century estimates, the historical data were used.
<table>
<thead>
<tr>
<th>Climate</th>
<th>Layer</th>
<th>Units</th>
<th>Resolution</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Current Climate</strong></td>
<td>Monthly Precipitation</td>
<td>Mm</td>
<td>100m</td>
<td>Hamann and Wang (2008)</td>
</tr>
<tr>
<td></td>
<td>Monthly Minimum Temperature</td>
<td>°C (* 10)</td>
<td>100m</td>
<td>Hamann and Wang (2005)</td>
</tr>
<tr>
<td></td>
<td>Monthly Maximum Temperature</td>
<td>°C (* 10)</td>
<td>100m</td>
<td>Hamann and Wang (2005)</td>
</tr>
<tr>
<td></td>
<td>Shortwave incoming radiation</td>
<td>Mj m2 day-1</td>
<td>100m</td>
<td>Kumar et al 1997</td>
</tr>
<tr>
<td><strong>2020 Climate</strong></td>
<td>Monthly Precipitation</td>
<td>mm</td>
<td>1km</td>
<td>Special Report on Emission Scenarios (SRES) climate scenarios (IPCC, 2000)</td>
</tr>
<tr>
<td></td>
<td>Monthly Minimum Temperature</td>
<td>°C (* 10)</td>
<td>1km</td>
<td>A2 scenario</td>
</tr>
<tr>
<td><strong>2050 Climate</strong></td>
<td>Monthly Maximum Temperature</td>
<td>°C (* 10)</td>
<td>1km</td>
<td></td>
</tr>
</tbody>
</table>

**2080 Climate**

<table>
<thead>
<tr>
<th>Climate</th>
<th>Layer</th>
<th>Units</th>
<th>Resolution</th>
<th>Reference</th>
</tr>
</thead>
</table>

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4-2. Develop methods over prototype study area, base level simulations

Figures 42 to 44 show an example of base level simulations acquired over the base study area. Figure 42 shows a digital elevation model and a Landsat composite acquired at the study site. Two examples for the cumulative precipitation on a monthly basis is shown in Figure 43. Precipitation during the winter month largely focused on the Rocky Mountain range with little precipitation in the eastern study area, whereas the rainfall was somewhat more equally distributed during the summer months. Figure 44 A-D shows a comparison between current minimum and maximum temperature in the study area in January and July.

Figure 42. Digital elevation model (A) and Landsat composite (B) of the study site
Figure 43. Average current precipitation in January (A) and July (B)
Figure 44. Current minimum and maximum temperature in the study area in January and July
CHAPTER 6: BEAR USE OF WHITEBARK PINE SEEDS AS A FOOD SOURCE IN THE WILLMORE WILDERNESS PARK 2008 PILOT STUDY

Tracy McKay¹ and Karen Graham¹

¹Foothills Research Institute

Introduction
The Willmore Wilderness Park in West Central Alberta falls within grizzly bear range, and is near the northern limit of whitebark pine trees (*Pinus albicaulis*) in the Canadian Rockies (Figure 1).

Figure 1: Eastern extent of grizzly bear range in Alberta, including the Willmore Wilderness Park.
The Foothills Research Institute (FRI) Grizzly Bear Program (GBP) has been conducting field activities near the Willmore in the Kakwa region, including collaring of bears, investigation of bear food distributions, and denning sites. The Willmore was also part of the DNA grizzly bear population census carried out in 2008. In the spring of 2008, the GBP received a request from the Parks Division of Alberta Tourism, Parks and Recreation to initiate a study in the Willmore, in order to investigate possible grizzly bear use of whitebark pine seeds as a food source. This request was based upon the following considerations:

1. The status of whitebark pine (endangered),
2. Current threats to whitebark pine in Alberta (primarily mountain pine beetle and white pine blister rust),
3. The presence of relatively large and healthy whitebark populations in the Willmore,
4. The well-documented relationship between grizzly bears and whitebark pine in the Greater Yellowstone Ecosystem, and
5. The need for a better understanding of the ecological significance of whitebark pine as a keystone species.

(Joyce Gould, personal communication, January 7, 2009).

In the United States, whitebark pine (WBP) seeds have been documented as a major source of energy for bears in the Greater Yellowstone Ecosystem (GYE) and the east slopes of the Montana Rocky Mountains (Mattson et al., 2001). WBP seeds are large and have a high fat content; Mattson et al. (1991) reported that when whitebark pine seeds were available and abundant, grizzly bears in the GYE ate almost nothing else; however, the use of whitebark seeds was subject to large-scale variation between years. Other studies have also reported both extensive and variable use of pine nuts by bears in the GYE (Kendall, 1983).

Whitebark pine trees are found at high elevations, from tree line down into the subalpine forest, and often grow on dry, south or southwest-facing slopes. In Alberta, WBP are distributed from the Kakwa Wildland Park in the north down the continental divide through Jasper and Banff national parks, to the southern edge of Waterton Lakes National Park at the U.S. border. The actual stands are poorly mapped in Alberta; therefore, we have a limited understanding of the distribution and abundance of Alberta’s WBP populations. However, it appears that the Willmore contains a relatively large and healthy whitebark pine population compared to other areas of Alberta (J. Gould, pers. comm., January 7, 2009).

In Canada, limited research has been completed on grizzly bear use of whitebark pine, and WBP seeds have not been previously reported as a significant food source. A food and habitat study in Banff National Park did not detect WBP seeds as a food item (Hamer and Herrero, 1987). Similarly, WBP was not reported as a food item in a study completed in Jasper National Park (Russell et al., 1979), or in an analysis of scat collected on the east slopes of West Central Alberta (Munro et al., 2006). None of the above study areas specifically considered whitebark pine stands, although the home ranges of some bears in the West Central Alberta study (Munro et al., 2006) may have
Chapter 6  
Whitebark Pine

extended into WBP distribution. There may be some overlap between WBP distribution in Alberta and existing grizzly bear location data. However, at this time, the lack of comprehensive WBP mapping makes it difficult to interpret bear use of WBP habitat based on GPS locations.

Grizzly bears are known to consume a wide variety of foods, and foods consumed vary by region. Therefore, due to the abundance of whitebark pine in the Willmore, and the lack of specific research investigating bear foods and use of WBP seeds in this area, it was conceivable that a relationship could exist between WBP and grizzly bears, similar to that observed in Yellowstone.

Previous studies in the U.S. have shown that all WBP seeds eaten by bears are obtained when bears excavate red squirrel middens; WBP cones are cached by squirrels after the cones have matured (Mattson and Reinhart, 1997). In areas of the GYE, WBP stands coincide with the upper-elevational limits of red squirrel distribution (Streubel, 1989, as cited in Mattson and Reinhart, 1997), and whitebark pine distribution overlaps with only a small part of squirrel range (Lorenz et al., 2008). At the start of this study, it was unconfirmed whether red squirrels inhabit the high elevation WBP stands in the Willmore. Since the consumption of WBP seeds by grizzly bears is dependent on red squirrels, an investigation of the potential availability and bear use of WBP seeds required a survey of red squirrel activity in WBP stands.

Based on the request from Alberta Parks, and upon a review of previous research in the U.S. as cited above, FRI researchers developed a set of basic study questions as a focus for the 2008 WBP pilot project in the Willmore. These questions were:

1) Are WBP seeds available as a food source for bears in the Willmore?
   a. What is the density (abundance) of WBP trees in the known Willmore WBP stands?
   b. Do red squirrels build middens in the WBP stands in the Willmore?
   c. If so, what is the midden density?
   d. Do squirrels cache significant numbers of WBP cones in WBP stands?

2) Are bears using WBP seeds as a food source in the Willmore?
   a. Is there evidence of bear activity and WBP use at squirrel middens?
   b. If so, how much? What is the relative importance of WBP in the diet?
   c. Is there seasonal and annual variation?

Based on preliminary observations in the field, additional research questions evolved, including examining potential relationships between squirrel midden densities and stand characteristics in the Willmore. Answering these research questions could provide further information on WBP seed availability for grizzly bears, and data collection methods were adapted to provide preliminary data to address these questions.

**Study area**
The study area included whitebark pine stands located within the Willmore Wilderness Park in west central Alberta (Figure 1). Point data of known WBP locations provided by
Alberta Parks and Alberta Sustainable Resource Development (ASRD) were used to delineate different study regions. Study regions were selected based on the size of the WBP stands (large stands were chosen) and areas that covered the spatial extent of the Willmore. Within the stands sampled, aspect ranged from southeast to southwest, slope angle averaged around 25 degrees, and the elevation was mainly between 1700m and 1900m. All WBP stands were located in mixed subalpine forests with various proportions of Englemann spruce and subalpine fir; occasionally, white spruce were also present. Canopy cover along transects ranged from 1 to 90%.

Methods
Fieldwork was completed during June 26-29, August 15-18, and September 26, 2008, and took place from fly-in camps and helicopter landing sites based near or within known whitebark pine stands. Transect surveys were completed to search for squirrel middens, estimate midden densities, complete midden surveys, estimate WBP densities, and to investigate grizzly bear activity at squirrel middens. Some sites (middens) were sampled again on repeat visits, while installing or taking down remote cameras.

Midden transect surveys:
Originally, we planned to use random points in WBP stands as the starting points for midden survey transects, in order to maximize scientific objectivity. However, areas (polygon spatial data) of WBP stands were not comprehensively mapped in the Willmore; only point data were available. We used GIS processing to select areas within the elevation range of WBP in regions where known WBP stands occurred. Random study sites were generated within these areas. Once in the field, it was obvious that this method was not effective, as the random points were not necessarily within actual WBP stands on the ground. In order to complete the maximum number of transects within WBP stands, researchers located WBP stands by observations on the ground, then traveled to the upper or lower elevation limits of the observed WBP stands, and started midden survey transects at a horizontal break (e.g. a streambed or gully) or at the beginning of a stand.

Data collected at the transect start included UTM coordinates, slope, aspect, elevation, percent tree canopy cover (estimated), shrub, forb and groundcover species present, a prism sweep of all tree species, and a general description of habitat.

From the transect start, a compass bearing was selected perpendicular to the direction of the slope (i.e. following the contour of the slope). Based on a sightability distance of approximately 5m along the ground in each direction, three researchers spaced themselves apart at a distance of 10m (one researcher was located at the transect start point, one was 10m upslope, and one was 10m downslope). In this way, a 30m wide area was searched for middens. Researchers followed the designated compass bearing across the slope, moving at a speed that allowed for detection of all middens within that range, until an obstacle or significant change in the landscape occurred. Transect lengths ranged

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2 A prism sweep is a forestry technique used to estimate basal area (cut stem area at ~1.4m) of tree species present, and is an indicator of tree density or productivity. The number of trees >/=1.4m tall are counted in the prism sweep, then multiplied by the prism factor to estimate the basal area of trees present in m²/hectare.
from 155 to 431m. During the second fieldwork trip (August), a prism sweep of all tree species was completed every 50m along each transect.

A full midden survey was completed for all middens observed along each transect. General data collected at the midden included UTM coordinates, slope, aspect, elevation, percent tree canopy cover (estimated), and shrub, forb and groundcover species present. A prism sweep was completed at the midden centre as well as 50m away from the centre in all four cardinal directions. The midden was measured (length, width, depth), midden tree species (central tree) was recorded, any cones visible at the midden were speciated and counted, and cone scale samples (approximate volume of 125mL) were counted (or collected for later counting) from each of the four cardinal directions from the tree centre. The midden was defined as active or inactive, based on the presence/absence of observed squirrel signs such as squirrel presence, a squirrel heard rattling, fresh cone piles, tree clippings, or piles of fresh cone scales. The midden and surrounding area were investigated for signs of bear activity, such as diggings (midden excavations) or bear scat. Any scat observed was collected and/or recorded, and the estimated age of bear scat or activity was also recorded.

At the end of each transect, researchers moved 30m up or downhill within the WBP stand (to avoid transect overlap) and started a new transect back in the opposite direction from the first. This pattern was repeated back and forth across the slope until essentially the entire WBP stand on that slope had been sampled, or as time permitted. At helicopter-accessed sites (versus sites accessed from camp), only two transects were completed at each site due to limited flying time.

Scat analysis:
Due to time restrictions and logistical issues associated with helicopter transport, not all bear scat samples observed near middens were collected for lab analysis, but all scats were examined in the field, and descriptions were recorded. If all scats observed near a midden appeared to contain the same food items, only one representative sample was collected; additional samples were collected for those that differed in appearance. Samples were stored in a cooler while in the field, and frozen at the FRI office for later analysis. In the lab, scat samples were allowed to thaw, and were spread out in a tray and dissected to identify food items present. Positive identification of food items was assisted by comparison to plant and scat samples in the FRI reference collection; researchers also collected WBP seed coats and seed remains in the Willmore for comparison purposes. Percentages of food items present by volume were estimated using a defined grid. Values recorded were not corrected for digestibility.

Remote cameras:
Remote cameras were set up at repeat sampling sites, in order to try to get photos of bears excavating squirrel middens. These photos could potentially provide more direct evidence of bear use of WBP, and the timing of any photos captured could help

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3 Based on an estimated home range of 100m by 100m for a red squirrel in spruce forest, as indicated in the literature: densities of 1 to 2 squirrels per hectare (Wheatley et al., 2002) correspond with approximate home ranges of 5000 to 10000m² (100m x 100m).
researchers understand how bears use the middens (e.g. whether they travel through to check the middens for WBP, how long they stay at a midden and/or how many times they return to the same midden). During the August fieldwork trip, five cameras were placed at middens where evidence of previous bear activity (e.g. excavations, scat) had been observed. The cameras were retrieved on September 26, 2008, and all photos were reviewed for the presence of grizzly bears.

Data analysis:
The area of each transect surveyed (in hectares [ha]) was calculated using the number of researchers, the number of horizontal metres surveyed by each researcher along transects, and the transect length (e.g. 3 researchers, 10m apart, 200m transect = 6000m² = 0.6ha). Total areas of the different study regions sampled within the Willmore were calculated by summing the areas of all transects surveyed within each region; midden densities and excavated midden densities (middens/ha) were then calculated by study region. Prism sweep data for each tree species were converted to basal areas (basal area in m²/ha = count of trees in prism sweep x prism factor), and basal area values recorded at all distances along all transects within a study region were averaged by study region. Based on observations of differences in stand composition within study regions (i.e. from one transect to the next), and the low number of study regions (2) with prism sweep data, basal areas and midden densities were also calculated by transect (see Mattson & Reinhart, 1996). Individual basal area values recorded at midden centres and four points 50m from the midden were reported as averages across all middens, and were also calculated as averages within two of the study regions, for comparison against transect values. Midden sizes were calculated as areas (length x width); volumes were not calculated due to difficulties in accurately measuring midden depth in the field.

Sample sizes were not sufficient for statistical analyses for most parameters, however, sample size and observed differences were large enough for comparison of midden sizes between excavated and unexcavated middens. Based on a non-normal distribution (as interpreted from histograms) for midden size data, a statistical analysis was completed using the Mann-Whitney U test for non-parametric testing of two independent samples.

Results
Midden transect surveys:
Over the course of the two field trips in June and August, sixteen transects were completed in four different regions of the Willmore, and midden surveys were completed at two additional study regions (Figure 2). Transect surveys covered a total area of 12.9 hectares; the area covered per study region varied from 1.2 to 5.3ha.
Prism sweeps along transects were completed during the August fieldwork trip only; basal area values for WBP in study regions C and D were 7.33 and 3.16 m²/hectare, with WBP trees making up 38.2 and 23.2 % of the total basal area of the stands. Other tree species observed on transects included Engelmann spruce (Se), subalpine fir (Fa), and white spruce (Sw). See Table 1 for values for basal areas and stand composition (percentages of total tree species) for tree species observed.

Table 1: Basal area and percentage of total basal area for tree species observed.

<table>
<thead>
<tr>
<th>Study region</th>
<th>Basal area (m²/hectare)</th>
<th>Percentage of total</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>WBP</td>
<td>Se</td>
</tr>
<tr>
<td>C</td>
<td>7.33</td>
<td>1.78</td>
</tr>
<tr>
<td>D</td>
<td>3.16</td>
<td>1.86</td>
</tr>
</tbody>
</table>

Red squirrel middens were observed along half (8/16) of the midden transects completed. A total of fifteen different middens and two major food caches were observed. Of the seventeen middens/caches, fifteen were active, and two middens were inactive (no recent squirrel sign). Red squirrel midden density by study region ranged from 0.60 to 1.68 middens per hectare (mean = 0.96, 95% CI = 0.47 – 1.44 middens/ha) (Table 2).
Table 2: Midden densities, active midden densities, and excavated midden densities for the four study regions.

<table>
<thead>
<tr>
<th>Study region</th>
<th>Middens</th>
<th>Active middens</th>
<th>Excavated middens</th>
</tr>
</thead>
<tbody>
<tr>
<td>A</td>
<td>0.60</td>
<td>0.60</td>
<td>0.20</td>
</tr>
<tr>
<td>B</td>
<td>0.74</td>
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<td>0.00</td>
</tr>
<tr>
<td>C</td>
<td>0.80</td>
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<td>0.00</td>
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<tr>
<td>D</td>
<td>1.68</td>
<td>1.50</td>
<td>1.12</td>
</tr>
</tbody>
</table>

The majority (13/17) of middens/caches were based around Engelmann spruce trees, and the remainder (4/17) were based at whitebark pine. Averaged over all 17 middens observed, basal areas for tree species observed around the midden were as follows: WBP = 4.16 m²/ha (95% CI = 3.24 – 5.07), Engelmann spruce = 5.03 m²/ha (95% CI = 3.84 – 6.22), Subalpine fir = 14.34 m²/ha (95% CI = 11.5 – 17.2), and total basal area = 23.9 m²/ha (95% CI = 21.0 – 26.7). In the two study regions where prism sweeps were completed at 50m intervals along transects, basal areas at middens differed from overall basal areas in that region; in particular, Engelmann spruce densities may be higher at middens than in the study region overall (Table 3). However, these results are preliminary only, and statistical analyses have not been completed to confirm the significance of these results.

Table 3: Basal areas within overall study regions versus basal areas at middens.

<table>
<thead>
<tr>
<th>Study region</th>
<th>Basal area (m²/hectare)</th>
<th>WBP</th>
<th>Se</th>
<th>Fa</th>
<th>Sw</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td>C</td>
<td>Average across study region</td>
<td>7.33</td>
<td>1.78</td>
<td>10.22</td>
<td>0.00</td>
<td>19.33</td>
</tr>
<tr>
<td></td>
<td>Average at all middens in study region</td>
<td>7.20</td>
<td>3.60</td>
<td>9.60</td>
<td>0.00</td>
<td>20.40</td>
</tr>
<tr>
<td>D</td>
<td>Average across study region</td>
<td>3.16</td>
<td>1.86</td>
<td>11.02</td>
<td>0.05</td>
<td>16.09</td>
</tr>
<tr>
<td></td>
<td>Average at all middens in study region</td>
<td>4.55</td>
<td>4.41</td>
<td>14.27</td>
<td>0.50</td>
<td>23.73</td>
</tr>
</tbody>
</table>

Of the 9 transects for which prism sweeps were completed, no middens were observed along transects with low (<2.5 m²/ha) basal area values for Engelmann spruce (Figure 3). Again, these observations are preliminary; sample sizes were not sufficient for statistical analysis regarding potential relationships between midden density and either stand composition or basal area of tree species present, and no conclusions can be made at this time.
Intact whitebark pine cones were cached on the surface at only one out of the seventeen middens sampled (6%), but WBP cone scales (new and/or old) were present at fifteen (88%) of the middens. There was high variation in the number of current-year WBP scales counted; on average, WBP scales made up about 12.7% of the total current-year cone scales at the middens (95% confidence interval = 3.28 – 22.1%). It should be noted that old WBP scales (from previous years) were present in significantly higher amounts at a number of middens; old WBP scales were observed in proportions as high as 80% of the total old cone scales at some middens.

Midden sizes (length x width) ranged from 1.87m$^2$ to 100m$^2$, with an average area of 36.9m$^2$ (95% confidence interval = 25.7 – 48.1). Middens that bears had excavated were significantly larger than those that were not excavated: average midden size at excavated middens was 43.1m$^2$, versus an average size of 23.2m$^2$ at middens not excavated by bears (Mann-Whitney U test statistic = 14, p = 0.03). Sample sizes were too small to analyze predictive relationships between either stand composition or density of WBP and midden size; more data could allow for a more complete (statistical) investigation of these potential relationships.

Four of the 16 transects (25%) included middens with evidence of bear activity, along with an additional midden found near one of the camps (independent of transect surveys). Overall, ten of the seventeen (58.8%) middens/caches showed evidence of bear activity; 8 of these had both midden excavations and scat present, and two had scat only. The excavated midden density ranged from 0 to 1.12 middens/ha (Table 2). New and/or old WBP cone scales were present at all middens with evidence of bear activity. Sample sizes were too small to analyze predictive relationships between either stand composition or density of WBP and midden excavations by bears; more data could allow for a more complete (statistical) investigation of these potential relationships.

All diggings and scat appeared to be from the previous year (2007) or older, and middens at repeat sample sites did not show any evidence of new bear activity between visits.
Scat analysis
At middens with evidence of bear activity, between 2 and 7 separate piles of bear scat containing WBP were observed per midden. A total of 38 separate bear scats were found at or near middens (Appendix 1); from field observations, WBP seed casings were visible in 35 of the 38 (92%). As previously described, if all scats observed near a midden appeared to contain the same food items, only one representative sample was collected for lab analysis. For all groups of scats described as having WBP visible in field observations, analysis of representative lab samples confirmed that WBP was present, indicating that field identification of WBP in scat samples was accurate. Nine scat samples were collected in total, and lab analysis identified WBP seed remains (seed coats) in 8 samples; five of these were 100% WBP. Other major food items identified included grass, vegetation, and crowberries (*Empetrum nigrum*). Out of interest, one scat sample not associated with a midden was collected, and was found to contain hoary marmot remains.

Remote cameras:
No photographs of bears were recorded by any of the five remote cameras set up at squirrel middens during the period of August 15 to September 26, 2008.

Discussion
In the two study regions where transect prism sweeps were completed, the values obtained for basal areas of WBP (3.16 and 7.33 m²/ha) were in the range of those observed in the GYE in regions of documented bear WBP use. Mattson and Reinhart (1997) reported WBP basal areas ranging from 2.2 to 23.4 m²/ha in the Yellowstone area, although the highest probability of bear excavations of middens occurred in stands with WBP basal areas in the range of 7.1 – 23.4 m²/ha (Mattson & Reinhart, 1997). Data from year one of this study were not sufficient to allow for a full analysis of WBP basal area as a predictor of either midden densities or bear activity in WBP stands, but an increased number of study areas during the proposed 2009 field season may provide enough data to investigate these relationships.

The observation of squirrel middens along study transects confirms that red squirrels do inhabit the high elevation WBP stands in the Willmore. Midden densities (0.6 to 1.68 middens/ha) and active midden densities (0 to 1.50 middens/ha, Table 2) observed in this study were similar to those previously reported in white spruce and lodgepole pine forests in Alberta (0.7 – 3.0 squirrels/ha, Wheatley et al., 2002), but higher overall than those reported in other studies of WBP stands (0.2 to 0.8 per hectare in the GYE, Mattson & Reinhart, 1996). These differences in midden densities could simply be due to the small sample size in our study (n = 4 for midden densities calculated per study region); densities observed may not accurately represent the study area as a whole. Midden density values may also have been affected by unequal sampling effort; the total area sampled was not identical for each study region, and the number of transects sampled was much lower than that surveyed by Mattson and Reinhart (1996). Alternately, the observed midden densities could reflect differences in stand composition between the Willmore and the GYE. Stand composition was a predictor of active midden densities in...
two study areas in the Yellowstone ecosystem (Mattson & Reinhart, 1997). The GYE includes a mix of lodgepole and whitebark pine, as compared to the mix of whitebark pine, spruce and fir found in the Willmore. Results from a study of red squirrels in Alberta suggested temporal changes in squirrel densities in different habitat types; squirrel densities were higher in spruce than in lodgepole pine, except in years following spruce cone failure (Wheatley et al., 2002). Other authors report that squirrels usually inhabit stands with multiple conifer species, as opposed to pure whitebark pine stands (Lorenz et al., 2008). Basal areas (densities) of dominant tree species and stand composition (proportions of different species) were considered in this study as possible predictors of squirrel midden density, and initial results suggest some relationships may exist. However, sample sizes were not sufficient for a full statistical analysis of potential relationships between midden density and either stand composition or tree densities; therefore, no conclusions can be made at this time. Further study of midden densities in WBP stands could help determine how stand characteristics may affect squirrel densities, and therefore help to characterize those WBP stands that are more likely to provide bears with WBP seeds. Increased sample sizes and more intensive sampling in additional WBP stands would also allow for more accurate comparison of stand characteristics, midden densities, and potential WBP seed availability between different study regions.

Based on evidence that bears obtain WBP seeds from squirrel middens, it follows that the number of WBP cones cached by squirrels would also affect the availability of WBP seeds for bears. The vast majority of middens did include at least some WBP cone scales, confirming that squirrels are collecting and using WBP cones in the Willmore, potentially making WBP seeds available for bears. However, very few intact WBP cones were cached at the middens observed in 2008. Availability of WBP cones was likely low in the Willmore in 2008; there appeared to be a cone crop failure at the sites studied (Dr. Vernon Peters, pers. comm., February 5, 2009). Whitebark pine trees produce mass cone crops (high seed production) every three to five years, with very low or no seed production in between (Morgan & Bunting, 1992, as cited in Alberta Sustainable Resource Development [ASRD] and Alberta Conservation Association [ACA], 2007). It is assumed that the inter-annual variability in cone production affects the number of WBP cones cached by squirrels, and therefore the number of WBP seeds available for grizzly bears.

It is important to note that only surface cones were counted in this study; middens were not disturbed to investigate buried cones. Any WBP cones from 2008 or previous years cached under the midden surface were not detected by our methods, and WBP cone caching by squirrels was likely somewhat underestimated. It is difficult to determine true WBP seed availability without digging up middens, which are crucial to winter survival for squirrels. Therefore, although the number of cones cached would provide the most direct estimate of WBP seed availability for bears, cone counts may be an appropriate surrogate measure of availability without disturbing middens. Similarly, Mattson (1994, as cited in Mattson et al., 2001) measured cones per WBP tree in relation to bear use of seeds.
Based on documented bear activity (excavations and scat) at squirrel middens, and direct evidence of WBP seed consumption by bears (presence of WBP seed casings in scat), results indicate that bears are using WBP seeds as a food source in the Willmore area. Excavations may represent the total consumption of WBP seeds by bears (Mattson & Reinhart, 1997); therefore, excavated midden densities may be used to estimate and compare levels of bear use of WBP seeds. Excavated midden densities varied among the four study regions in this study (from 0 to 1.2 excavated middens/ha), suggesting that bear use of WBP may differ between the four regions. Mattson and Reinhart (1997) reported a range of 0 to 0.291 excavated middens/ha in the GYE. As described for active midden densities, the higher values for excavated midden densities in this study could be a result of the small sample size in our study, or inconsistent sampling effort. Alternatively, higher numbers of excavated middens could reflect a higher midden density supported by the stand composition in the Willmore. Regardless, values for both active midden densities and excavated midden densities are in the approximate range of previously reported values, and the presence of excavated middens confirms that squirrel midden densities are adequate to support bear use of WBP seeds.

The presence of multiple scat piles observed near middens may indicate that bears return to the same midden, and/or remain at middens for several feeding sessions. Other research has also reported repeat excavations of squirrel middens over the course of months (Kendall, 1983).

WBP basal area was a predictor of excavations by bears in the GYE (Mattson & Reinhart, 1997). As with squirrel midden densities, sample sizes in this study were not sufficient for a full statistical analysis of potential relationships between excavated midden densities and either stand composition or tree densities. Increased sample sizes and more intensive sampling during future (proposed) research in additional WBP stands would allow for further comparison of levels of bear use (excavated midden densities) and stand characteristics between different regions of WBP distribution, and investigation of potential predictor variables for levels of bear activity in WBP stands.

Data suggest that a relationship exists between midden size and excavations by bears; excavated middens were significantly larger than middens not excavated by bears in this study. Mattson and Reinhart (1997) also reported that larger middens were excavated more often than smaller middens. Based on these results, midden size is one of a number of factors that could be used to predict bear use of WBP seeds in different areas of WBP distribution.

Observations suggest that bear use of WBP seeds did not occur in 2008 in the study area: all cases of documented bear use of WBP seeds were from previous years, no new bear sign was observed during the course of the study, and remote cameras did not obtain any photos of bears at squirrel middens. As mentioned, availability of WBP seeds was likely low in the Willmore area in 2008, as indicated by the low number of WBP cones observed on WBP trees and the low number of cones cached at squirrel middens. Other authors have also noted inter-annual variation in WBP use, related to the inter-annual variation in WBP cone production. In a ten-year study of food habits of Yellowstone
grizzly bears, use of WBP seeds varied substantially between years, including years with no use as well as years when WBP seeds made up most of the food consumed in summer and fall (Mattson et al., 1990). Kendall (1993) reported annual variation in bear use of pine nuts, depending on availability (timing of cone maturity and squirrel caching) and abundance (peak cone crop years). Mattson et al. (2001) report a relationship between grizzly bear consumption of WBP seeds and the size of the cone crop, suggesting that there may be a minimum threshold of availability of WBP seeds for bear use to occur.

The presence of ripe berries in conjunction with WBP seed casings in two scat samples suggests fall use of WBP seeds, although it was not possible to determine the exact age of the scat observed. Seasonal variation of WBP was noted in food habits of Yellowstone bears, with the highest use in September and October (Mattson et al., 1990). Kendall (1983) reported that spring use of WBP seeds in the GYE correlated with cone production from the previous year, and fall use was correlated with current year’s crop. Based on the above information and the lack of activity in 2008, bear use of WBP observed in this study is assumed to be fall use of the 2007 cone crop.

WBP seed casings were observed (and confirmed by lab analysis) in almost all (92%) of the bear scat samples found near excavated middens. However, it is difficult to determine the relative importance of WBP seeds in the overall diet of grizzly bears in the Willmore, since sample collection efforts were not temporally and spatially random, or equally distributed across the landscape (scat samples were recorded and collected only along transects). In order to estimate relative importance of a food item, multiple scat samples must be collected from the same (collared) bear (e.g. Munro et al., 2006); this was not possible during our study. Regardless, although we could not determine what proportion of the overall or seasonal diet was composed of WBP seeds, the presence of WBP seed casings in numerous bear scats does confirm that bears in the Willmore are using WBP seeds as a food source of some importance. In addition, relative levels of use in different regions of WBP distribution may be inferred and compared using excavated midden densities; as discussed, excavated midden densities varied among the four study regions in this project, suggesting that bear use of WBP may differ between the four regions. A grizzly bear population estimate (based on a DNA census) for this area will be available in spring of 2009; information regarding the number of bears in the Willmore area will provide more background on the relative importance of whitebark pine habitat to the overall grizzly bear population in Alberta.

Previous research in Alberta has not reported WBP seeds as a significant food source for grizzly bears, however, as previously discussed, research did not focus on areas within WBP distribution. Although WBP seed availability and seed use by bears may be localized, and may change significantly from year to year, WBP seeds could still serve as an important energy source for grizzly bears in areas of WBP distribution. Bears are known to be generalists, consuming a diverse range of foods, depending on seasonal, annual, and regional availability (Munro et al., 2006). With regards to WBP seeds and other variable food sources, Mattson et al. (1991) refer to the role of “episodically significant periods of food”, particularly for a “marginally viable population”; food items used infrequently may still be of great importance during the years of heavy use. The
high fat content of WBP seeds makes them a high quality food item, both due to the calories they contain and the efficiency with which they can be converted to body fat (Mattson et al., 2001). Female bears have been reported to eat twice as many pine seeds as males; reproductive success and lactation of females depends upon fat reserves, and consumption of high fat WBP seeds may facilitate the accumulation of body fat (Mattson et al., 2001). In the GYE, there is also a strong relationship between whitebark pine seed crop size and grizzly bear survival (Mattson et al., 1992). Use of WBP seeds during years of peak cone crops (mast years) could affect grizzly bear survival and reproduction in the Willmore and other areas of WBP distribution in Alberta.

Conclusions
Fieldwork conducted in 2008 was intended as exploratory research, as the relationships among whitebark pine, red squirrels and grizzly bears had not previously been documented in Alberta. As such, data obtained are preliminary, and sample sizes were not sufficient for extensive statistical analyses. However, 2008 data confirm that grizzly bears in the Willmore Wilderness use squirrel-cached WBP seeds as a food source. This information could be useful for grizzly bear food models in areas of WBP use, and is a consideration in assessing quality of bear habitat.

Current threats to whitebark pine in Alberta include white pine blister rust, mountain pine beetle, climate change, and prolonged fire suppression (ASRD & ACA, 2007), and the status of WBP in Alberta has recently been changed to endangered. The forest changes resulting from mountain pine beetle infestation are being considered in current Grizzly Bear Program research (see FRI, 2007); establishing relationships between bears and whitebark pine may provide more information on the potential impacts of mountain pine beetle on grizzly bear populations. Results from this study contribute to a better understanding of the ecological significance of whitebark pine. If whitebark pine seeds are a significant grizzly bear food, the potential loss of WBP trees through blister rust or pine beetle could affect the reproduction and survival of grizzly bears in areas of WBP distribution. WBP distribution includes a considerable area of grizzly bear range in Alberta, from the Kakwa Wildland Park south along the continental divide to the United States. As another species at risk, the connection of this endangered tree species to grizzly bears could make WBP conservation efforts even more of a priority in Alberta, and in certain areas, the coordinated management of the two species could be useful. There is a need for more complete and accurate mapping of WBP stands in Alberta; future work has been proposed by other researchers to address this knowledge gap. Improved knowledge of WBP distribution will assist in investigating grizzly bear use of pine seeds.

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4 WBP conservation efforts currently underway include improving WBP stand inventories, collecting seeds from blister rust-resistant trees, prescribed burning, forest health monitoring sites, and Verbenone trials (mountain pine beetle control) (ASRD & ACA, 2007).

5 Researchers from the University of Calgary, University of Alberta, King’s University College, University of British Columbia, Parks Canada, and Alberta Sustainable Resource Development (Forest Health and Parks Divisions) and FRI met in March to discuss ongoing and potential future WBP research; improved mapping of WBP distribution in Alberta was identified as a priority.
Other authors have suggested that grizzly bears would benefit from the minimization of roads and other human facilities in areas of whitebark pine distribution (Mattson & Reinhart, 1997); based on known relationships between grizzly bear mortalities and roads in Alberta, this suggestion may also be supported for the Alberta WBP zone. Much of WBP distribution is within protected areas in Alberta, with relatively low road densities. However, the southern extent of WBP distribution and other areas outside the parks have higher road densities, and road development will likely increase in unprotected areas. Currently, it is difficult to accurately measure road densities within WBP stands in Alberta, due to incomplete mapping of WBP, but roads and development in areas of WBP could be a consideration in bear management.

Further work is needed to develop this research, and to expand the study area to include more regions within WBP distribution in west-central Alberta. Collecting more data will allow better understanding of relationships among whitebark pine, squirrels and bears, of the relative importance of WBP seeds for grizzly bears, of potential differences between different regions of WBP distribution in terms of availability of WBP seeds and levels of bear use, and of possible predictor variables for WBP availability and levels of bear use. Different regions within WBP distribution may support different levels of bear use of pine seeds; information regarding areas of high levels of grizzly bear WBP use could assist in prioritization of WBP conservation efforts and/or influence bear conservation strategies in areas within WBP distribution.

**Literature cited**


CHAPTER 7: AVAILABILITY OF BEAR FOODS IN THE CHINCHAGA REGION: FIELD DATA COLLECTION FOR GRIZZLY BEAR FOOD-HABITAT MODELS

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¹Foothills Research Institute

Introduction
The Chinchaga region includes grizzly bear habitat in northern Alberta, from west of the town of Manning to the British Columbia border (Figure 1). This area was part of the Phase 7 extension of the remote sensing landcover mapping of grizzly bear range in Alberta, completed in 2007/2008. Due to difficulties in collaring bears in the region, there is a lack of GPS collar data for the Chinchaga, and a tested and validated Resource Selection Function (RSF) map cannot be produced for this area. Food-based habitat mapping is a potential alternative to radio telemetry-based habitat mapping; food based models address limiting factors (food resources), and do not require capturing or collaring of bears (See Chapter 8 & Nielsen, 2008). Landcover type is used as one of the predictors of bear food distribution in food-based habitat models;⁶ field data regarding the distribution and abundance of bear foods in different landcover types is required.

Figure 1: Overview of the Chinchaga area.

⁶ Additional parameters applied in food models include climate, terrain, and forest stand characteristics.
Research objectives:
To provide preliminary data for a food-based habitat model for the Chinchaga, by determining, (within different land cover types):
1. Availability of known bear foods (presence/absence),
2. Levels of abundance (percent cover) of bear foods, and
3. Forest stand characteristics.

Study Area
The Chinchaga 2008 study area includes the Lower Foothills and Upper Foothills ecosite subregions, and encompasses four Forest Management Agreement areas (FMAs), as well as the Chinchaga Wildland Provincial Park. Two of the FMAs in the region are managed by Daishowa-Marubeni International (DMI), and two by Manning Diversified (Figure 2).

![Figure 2: Chinchaga 2008 study area, including Forest Management Agreement areas and the Chinchaga Wildland provincial park.](image-url)
The region borders on agricultural lands, and the vast majority of the study area is forested, dominated by broadleaf forests and treed wetlands, along with conifer and mixed forests (Figure 3).

Figure 3: Landcover types in the Chinchaga study area.

Methods
Site selection
Study area delineation:
The 2008 study area was adapted from the 2006 and 2007 FRI bear capture areas, and was drawn to encompass most of the following points: FRI collar points from 2 collared bears, SRD collar points (1999-2001), known mortality locations (1988-2006), and reported sightings from 2008 (from Daishowa-Marubeni International [DMI]). Some collar points were excluded from the study area, due to location within agricultural lands, and based on knowledge that the outlying points were from a relocated bear (i.e. not within its home range). The northern extent of the study area follows the available land cover raster data, and some parts of the study area polygon were diverted around towns.
**Landcover types:**

Landcover polygons for the Chinchaga area were created from the raster (remote sensing) coverage available from the GBP phase 7 deliverables. It was determined (based on input from Karen Graham, FRI and Scott Nielsen, University of Alberta) that landcover types most important for food availability (shrub, herbaceous, mixed forest, and broadleaf forest) should be separated out into riparian and non-riparian areas, since riparian areas hold additional importance for bears. Dense conifer forest and moderate conifer forest were combined to reduce the number of landcover classes, and open conifer forest was excluded as a target landcover due to its extremely low distribution (<1% of the total study area). Open wetlands were also excluded, since they are not bear habitat, and no study sites were sampled in agricultural areas. It was determined that cutblocks are an important landcover type, and should be distinguished from the other landcover classes (cutblocks were not an individual landcover class in the raster coverage). Cutblock data (polygons) for the Chinchaga area were obtained from SRD for the years up to 2006. Cutblocks were separated into age classes based on input from Scott Nielsen, and modified based on input from Jim Witiw, DMI (regarding possible faster succession in the Chinchaga). Based on the process described above, there were fifteen landcover types used to define study sites (Table 1).

**Table 1: Landcover types used for study site selection.**

<p>| | |</p>
<table>
<thead>
<tr>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>shrub non-riparian</td>
</tr>
<tr>
<td>2</td>
<td>herb non-riparian</td>
</tr>
<tr>
<td>3</td>
<td>mixed forest (MF) non-riparian</td>
</tr>
<tr>
<td>4</td>
<td>broadleaf forest (BL) non-riparian</td>
</tr>
<tr>
<td>5</td>
<td>shrub riparian</td>
</tr>
<tr>
<td>6</td>
<td>herb riparian</td>
</tr>
<tr>
<td>7</td>
<td>MF riparian</td>
</tr>
<tr>
<td>8</td>
<td>BL riparian</td>
</tr>
<tr>
<td>9</td>
<td>Dense conifer/Moderate conifer forests (DC/MC)</td>
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<tr>
<td>10</td>
<td>Treed wetlands (TW)</td>
</tr>
<tr>
<td>11</td>
<td>Cutblocks, 2-5 years post-harvest</td>
</tr>
<tr>
<td>12</td>
<td>Cutblocks, 6-10yrs</td>
</tr>
<tr>
<td>13</td>
<td>Cutblocks, 11-20yrs</td>
</tr>
<tr>
<td>14</td>
<td>Cutblocks, 21-30yrs</td>
</tr>
<tr>
<td>15</td>
<td>Cutblocks, &gt;30 years</td>
</tr>
</tbody>
</table>

**Generating random sampling points:**

Hawth’s tools⁷ were used to generate random points within each landcover type. The minimum distance between sampling points was set at 500m to attempt an even distribution across the entire study area, and an unstratified sampling design was applied (due to the nature of the multi-part landcover polygons). Points were separated into road-accessible and helicopter-accessible points; road accessible points were defined as those within 500m from the nearest (usable) road; heli-accessed sites were those >500m from

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⁷ Hawth’s Analysis Tools for ArcGIS are a well known GIS toolset used for ecology research; see http://www.spatailecology.com/htools/tooldesc.php.
the nearest road or access point (e.g. pipelines). Due to cost and logistics, the majority of points were accessed by road. The number of study sites sampled in each landcover type was approximately proportional to the percentage of the study area comprised by each landcover (Table 2), except for those excluded as described above (see B, Landcover types) and in the case of cutblocks, where an increased number of plots were sampled to capture different regeneration ages.

Table 2: Areas of landcover types in the Chinchaga, and number of study sites sampled.

<table>
<thead>
<tr>
<th>Landcover</th>
<th>Area (km²)</th>
<th>%</th>
<th># sample plots</th>
</tr>
</thead>
<tbody>
<tr>
<td>Agricultural</td>
<td>384</td>
<td>1.62</td>
<td>0</td>
</tr>
<tr>
<td>Barren Land</td>
<td>59</td>
<td>0.25</td>
<td>0</td>
</tr>
<tr>
<td>Broadleaf forest</td>
<td>4734</td>
<td>19.96</td>
<td>11</td>
</tr>
<tr>
<td>Cutblocks (all ages)</td>
<td>1206</td>
<td>5.09</td>
<td>20</td>
</tr>
<tr>
<td>Dense or moderate conifer</td>
<td>3798</td>
<td>16.02</td>
<td>9</td>
</tr>
<tr>
<td>Herb</td>
<td>949</td>
<td>4.00</td>
<td>2</td>
</tr>
<tr>
<td>Mixed forest</td>
<td>5246</td>
<td>22.12</td>
<td>9</td>
</tr>
<tr>
<td>No data</td>
<td>0.8</td>
<td>0.00</td>
<td>0</td>
</tr>
<tr>
<td>Open conifer</td>
<td>58</td>
<td>0.24</td>
<td>2</td>
</tr>
<tr>
<td>Open wetland</td>
<td>329</td>
<td>1.39</td>
<td>0</td>
</tr>
<tr>
<td>Shrub</td>
<td>2447</td>
<td>10.32</td>
<td>6</td>
</tr>
<tr>
<td>Treed wetland</td>
<td>4389</td>
<td>18.51</td>
<td>9</td>
</tr>
<tr>
<td>Water</td>
<td>116</td>
<td>0.49</td>
<td>0</td>
</tr>
<tr>
<td><strong>Total</strong></td>
<td><strong>23717</strong></td>
<td><strong>100</strong></td>
<td><strong>68</strong></td>
</tr>
</tbody>
</table>

Field methods
Field methods were adapted from previous microsite data collection methods. Researchers navigated to the random point selected, then established a 30m by 30m study plot centred on the random point. General data collected at study sites included elevation, slope, aspect, anthropogenic features, and general stand ecology. Stand characteristics recorded included basal area (prism sweep), stand seral stage, distance to edge (if applicable), percent canopy cover (estimated), average height and DBH for dominant and subdominant tree species, average height for dominant and subdominant shrub species, and abundance of coarse woody debris and hanging woody debris. Wildlife sign (pellets, tracks, trails, browsing) was recorded along with any bear microsite use or activity (anting, bedding, etc.). Ant mounds or other insect nests (e.g. yellow jackets) were counted. Soil textures were classified at each site, and depth of humus layer was recorded.
Plant species were classified as bear foods based on previous microsite data collection, foods directly observed in bear scat in the Chinchaga region, and information found in the literature for plant species occurring in the area (see Appendix 1 for a list of bear food plant species). Study plots were searched for the presence of any bear foods, and all bear food plant species present were assigned a percent cover value (Table 3). Presence and abundance (rare or common) of fruit was recorded if applicable, and any fruit present was classified as unripe, ripe, or rotten.

### Table 3: Ranges for percent cover values

<table>
<thead>
<tr>
<th>Rank</th>
<th>Percent Cover</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>&lt; 1% or ~1 to 5 individuals</td>
</tr>
<tr>
<td>2</td>
<td>&lt; 5% or ~6-50 individuals</td>
</tr>
<tr>
<td>3</td>
<td>5-25%</td>
</tr>
<tr>
<td>4</td>
<td>26-50%</td>
</tr>
<tr>
<td>5</td>
<td>51-75%</td>
</tr>
<tr>
<td>6</td>
<td>76-100%</td>
</tr>
</tbody>
</table>

In addition to bear foods, presence/absence of indicator plant species and ecosite phase classification were recorded.

**Scat collection and analysis**

Any bear scat samples observed en route to study sites, along roads, or at study plots were collected for analysis. Samples were stored in a cooler while in the field, and frozen at the FRI office for later analysis. In the lab, scat samples were allowed to thaw, spread out in a tray, and dissected to identify food items present. Positive identification of food items was assisted by comparison to plant and scat samples in the FRI reference collection; researchers also collected local food samples from the Chinchaga area for reference. Percentages of food items present by volume were estimated using a defined grid. Values recorded were not corrected for digestibility. Samples were also sent for DNA testing to distinguish black bear scat samples from grizzly bear scat, however, results were not yet available at the time of publication.

**Results**

**Vegetation sample plots:**

A total of 68 study plots were sampled in the 15 different landcover types defined for study sites (see Tables 1 and 2). Forest stand characteristics, bear food presence, and percent cover (abundance) for bear food plant species are included in the food-based habitat model currently being developed for the Chinchaga area as part of our 2008 deliverables.

**Scat analysis:**

Eleven scat samples were collected in spring (June), one in summer (July), and six in fall (September); all 18 scat samples were analyzed in the lab. Food items identified in scat included grass, clover (Trifolium sp.), ants and ant larvae, rose hips, currants and gooseberries (Ribes sp.), cranberries (Viburnum sp.), blueberries (Vaccinium sp.),

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8 As defined by the Field Guide to Ecosites of West-central Alberta (Beckingham et al., 1996).
dogwood berries (Cornus stolonifera), pea-vine vegetation (Lathyrus sp.), dandelion, horsetails (Equisetum), beaver, and vegetation of unknown species. Bear diets are diverse; this is reflected in the high variation in proportions of food items in scat, therefore occurrence of food items (percentage of scat samples containing food items) are presented here rather than mean volumes of food items (Figure 4).

![Figure 4: Occurrence of bear food items in scat collected in the Chinchaga region.](image)

Discussion
Scat results were consistent with seasonal use of food items, with more consumption of vegetation in the spring, and use of fruit in the fall. Use of food items also reflected differences in locally available bear foods as compared to other areas (e.g. West-Central Alberta, see Munro et al., 2006); there was more use of rosehips than expected, no buffaloberries were detected in scat, and use of cranberries was much greater than consumption of Vaccinium species. Rose and Viburnum (cranberry) are dominant species in the Chinchaga area, while Shepherdia (buffaloberry) was observed at fewer study sites. The presence of beaver meat in scat samples also reflects local availability of foods; beaver lodges are present in high numbers on the landscape in the Chinchaga region.
As previously described, scat collected during 2008 was sent for analysis to distinguish black bear scat samples from grizzly bear scat samples, but results were not available at the time of publication. Food items identified in scat analysis therefore may have been consumed by black bears. In addition, sample collection efforts were not random, systematic, or equally distributed across the landscape (scat samples were collected en route to study plots). However, the scat analysis results do provide an overview of bear foods in the area. Information from the scat analysis will be used in the food based model to highlight the temporal importance of the various grizzly bear foods in the Chinchaga region.

**Literature cited:**


Appendix: Bear food plant species in the Chinchaga.

<table>
<thead>
<tr>
<th>Species</th>
</tr>
</thead>
<tbody>
<tr>
<td>Amelanchier spp.</td>
</tr>
<tr>
<td>Arctostaphylos uva-ursi</td>
</tr>
<tr>
<td>Empetrum nigrum</td>
</tr>
<tr>
<td>Equisetum spp.</td>
</tr>
<tr>
<td>Frageria spp.</td>
</tr>
<tr>
<td>Hedysarum spp.</td>
</tr>
<tr>
<td>Heracleum lanatum</td>
</tr>
<tr>
<td>Lonicera spp.</td>
</tr>
<tr>
<td>Medicago spp.</td>
</tr>
<tr>
<td>Prunus spp.</td>
</tr>
<tr>
<td>Ribes spp. (currants)</td>
</tr>
<tr>
<td>Ribes spp. (gooseberry)</td>
</tr>
<tr>
<td>Rubus spp.</td>
</tr>
<tr>
<td>Rubus idaea</td>
</tr>
<tr>
<td>Rubus parviflorus</td>
</tr>
<tr>
<td>Sambucus racemosa</td>
</tr>
<tr>
<td>Sheperdia canadensis</td>
</tr>
<tr>
<td>Taraxacum officinale</td>
</tr>
<tr>
<td>Trifolium spp.</td>
</tr>
<tr>
<td>Vaccinium caespitosum</td>
</tr>
<tr>
<td>Vaccinium membranaceum</td>
</tr>
<tr>
<td>Vaccinium myrtilloides</td>
</tr>
<tr>
<td>Vaccinium scoparium</td>
</tr>
<tr>
<td>Vaccinium vitis-idaea</td>
</tr>
<tr>
<td>Viburnum sp.</td>
</tr>
</tbody>
</table>
CHAPTER 8: A RESOURCE-BASED HABITAT MODEL FOR GRIZZLY BEARS (URSUS ARCTOS) IN WEST-CENTRAL ALBERTA, CANADA

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Department of Renewable Resources, University of Alberta

SUMMARY
1. Wildlife habitat modeling often relies on telemetry data and indirect resource variables. Such assessments frequently ignore animal behaviour and rarely act as an index of habitat quality. We describe a resource-based model of habitat quality that is based on the diet of the species and the distribution and seasonal availability of those resources. We demonstrate our model for a population of grizzly bears (Ursus arctos) in west-central Alberta, Canada and examine relationships to observed habitat use and regional occupancy-abundance.

2. Resource patches were predicted for root digging, herbaceous grazing, animal matter, and frugivory foraging activities using field plots and logistic regression. Potential habitat quality was estimated by weighting resource patches at bi-monthly periods based on seasonal diets. Habitat use of resource patch and potential habitat quality models were tested using 1032 field-visited telemetry locations from 9 grizzly bears with known foraging activity and 42853 telemetry locations from 44 grizzly bears respectively. Realized habitat quality, which accounted for range loss, was compared to occupancy-abundance at hair-snag sites.

3. Distribution of resource patches, as well as potential habitat quality, significantly predicted habitat use by grizzly bears, while regional patterns of occupancy-abundance were positively related with realized habitat quality. Habitat potential was high in montane mountain valleys where grizzly bears have been displaced by high mortality resulting in significant habitat deficits.

4. Synthesis and applications. Resource-based wildlife habitat models, which explicitly recognize adaptive switching strategies in animals, can predict habitat use and occupancy-abundance of wildlife populations. Since our approach does not require location data for estimation, it avoids many of the pitfalls associated with telemetry-based studies. We suggest that maps of habitat deficit be considered when prioritizing conservation actions for population recovery and that future resource models incorporate energetic measures, such as kilocalories or potential net digestible energy.

Introduction
Wildlife habitat modeling often relies on telemetry information to make inferences about habitat needs (White & Garrott 1990; Manly et al. 2002) and to map critical habitats for
management and conservation action (Boyce & McDonald 1999; Johnson et al. 2004; Nielsen et al. 2006). Most typically, habitat models pool telemetry locations and as a consequence animal activity. This assumes that the general distribution of animals is sufficient to define habitat, despite the plea by some that wildlife management and conservation necessitates more than just the knowledge of species distribution (e.g. Garshelis 2000). Specific activities or behaviours are certainly more likely to be critical or limiting to the population than others. For instance, food resources (nutrients/energy) are often a critical factor affecting population growth or abundance (Miyashita 1992; Mattson et al. 2004; Brasher et al. 2007). Our understanding and characterization of wildlife habitat could benefit from modeling efforts that emphasize foraging activities, rather than movement or resting activities, along with measures of connectivity (Chetkiewicz et al. 2006) or habitat loss/displacement due to human disturbance (Mattson & Knight 1991; Gibeau 1998; Mattson et al. 2004; Nielsen et al. 2006).

One approach to using telemetry data to define a resource-based habitat model is to classify behavioural activities, such as intra- and inter-patch movements (using movement rates), and to separately assess habitat associations for each category (e.g., Johnson et al. 2002a, 2002b). However, classification of movement rates from telemetry locations may be sensitive to sampling interval (Nams 2006; Bradshaw et al. 2007; Pinaud 2008) and location error (Bradshaw et al. 2007). While global positioning system (GPS) telemetry has greatly increased positional accuracy compared to prior technologies (e.g., very high frequency- VHF radiotelemetry; White & Garrott 1986; Nams 1989), acquisition failures due to terrain ruggedness and canopy cover are well-documented (Remple et al. 1995, Moen et al. 1996, D’Eon et al. 2002; Di Orio et al. 2003, Frair et al. 2004, Cain et al. 2005) and potentially troubling, since errors are often directional, and may be caused by the same factors used in subsequent wildlife-habitat modeling. Animal behaviour can further magnify acquisition biases (Bowman et al. 2000; Graves & Waller 2006; Heard et al. 2008). Recent technological advances in telemetry, such as cameras and/or video (Moll et al. 2007), activity sensors (Coulombe et al. 2006), or biotelemetry systems that measure physiology, behaviour, and energetic status (Cooke et al. 2004) may address many of these challenges. However, most of these technologies are still under development, and are not yet widely available in commercial systems.

Another criticism of wildlife-habitat and species-distribution models is that the environmental variables used to create them are often selected on the basis of availability rather than biological relevance (Austin & Smith 1989; Guisan & Zimmermann 2000; Austin 2007). For instance, elevation from a digital elevation model (DEM) is commonly used for predicting species distribution and habitat use. However, elevation acts as a surrogate or indirect variable for a suite of other factors influencing plant growth (Austin & Smith 1989), such as temperature (direct physiological variable) and moisture (resource gradient variable, albeit a distal variable; Austin 2007). While both of these factors vary with elevation, the relationship is inconsistent, even over small spatial scales. An over-reliance on such indirect variables is perhaps one reason why species-distribution and especially animal habitat-selection models extrapolate so poorly (Austin & Smith 1989; Johnson et al. 2008). We contend that an approach to wildlife habitat
modeling that explicitly targets mechanistic resources variables will be more valuable to scientists than models based on commonly available indirect factors alone (Austin 2007).

Even with the proper selection of relevant resource factors, animal habitat models that rely on telemetry-based assessments can only provide a measure of use (presence). Unused (absence) locations are unknown, and present an asymmetry in the response measure (Johnson et al. 2006); a condition which is also common to species distribution modeling based on herbarium or museum records (Zaneiwicki et al. 2002). Given the lack of unused or absent locations, a sample of random, available locations (pseudo-absences) is commonly used as a substitute. Environmental conditions are then compared among used and available locations to estimate species distribution or habitat selection (Hirzel et al. 2002; Manly et al. 2002; Zaneiwicki et al. 2002; Engler et al. 2004). The choice of spatial scale over which random observations are drawn, however, affects inferences (Boyce et al. 2006; Allouche et al. 2008). Temporal scale is also important to consider. For instance, many animals (especially generalists) have evolved an adaptive switching strategy (Ma & Levin 2006) that can be characterized by changing habitat use based on availability of resources (Schwemmer & Garthe 2008). Regardless of the spatial or temporal scale chosen, common analytical approaches to comparing random observations to used (present) observations, such as resource selection functions (Manly et al. 2002) or logistic regression (Zaneiwicki et al. 2002), may prove to be problematic due to contamination (Keating & Cherry 2004). In the case of telemetry-based observations of animal habitat use, contamination represents undetected habitat use of collared or unmarked animals. Keating & Cherry (2004) suggest that such errors are a serious flaw to resource selection functions (presumably also for methods using pseudo-absences in herbarium/museum records), although Johnson et al. (2006) demonstrated through simulations that habitat selection analyses are often robust to contamination levels below 20%, consistent with recommendations from Landcaster & Imbens (1996).

Given the challenges of defining animal activity, location errors and biases inherent in telemetry, choice of proper predictor variables, difficulties in defining or understanding scale, and problems associated with contamination, alternative approaches to wildlife-habitat modeling are needed. Ideally, any alternative would focus on critical animal behaviours that most affect population demography, explicitly recognize adaptive switching behaviours and temporally relevant scales, avoid errors and biases associated with telemetry, select direct or resource gradients for predictors, and avoid issues of contamination. In this paper, we suggest a general approach to wildlife habitat modeling and mapping that measures habitat quality as the sum of weighted resource items. The approach avoids many of the telemetry-based pitfalls by focusing on the identification of critical resources and their availability in space and time. We apply our habitat model to a population of grizzly bears (Ursus arctos) in west-central Alberta, Canada and verify its utility by examining the relationship between habitat quality and seasonal habitat use (telemetry) and regional patterns in occupancy-abundance. We also illustrate the estimation of realized habitat quality by considering range loss allowing quantification of a habitat deficit index. Such indices may be helpful in prioritizing areas for the recovery (restoration or reintroduction) of threatened species, such as grizzly bears.
Materials and methods

Study area

We estimated habitat conditions for a population of grizzly bears in west-central Alberta, Canada (53° 15' N 118° 30' W; Figure 1). The 8,606-km² study area encompassed the Rocky Mountain and Foothills natural regions (Natural Regions Committee 2006) near the town of Hinton (Figure 1). The Rocky Mountain region is characterized by a broad range of physical and environmental characteristics, including montane openings at lower altitudes, forests at intermediate altitudes, and alpine meadows, rock, and ice at the highest altitudes. Forest stands in the mountains are dominated by coniferous stands of lodgepole pine (*Pinus contorta*), spruce (*Picea englemanii* and *P. glauca*), and sub-alpine fir (*Abies lasiocarpa*). Land-use activities in most areas of the mountains are restricted to recreational pursuits, with most of the region protected by Jasper National Park and other small provincial reserves. In contrast, the Foothills region is managed for multiple land use, including forestry, oil and natural gas exploration and development, coal mining, residential areas, agriculture, and recreational activities. While land cover is generally dominated by forests, small natural openings (marshes, bogs, and low-gradient riparian areas), and agricultural fields are also present. Forests in the Foothills are comprised of widespread lodgepole pine on dry sites, black spruce (*Picea mariana*) and tamarack (*Larix laricina*) on wet sites (particularly at lower altitudes), and mixed stands of quaking aspen (*Populus tremuloides*), lodgepole pine, balsam fir (*Abies balsamea*) and white spruce (*Picea glauca*) on mesic sites at low elevations.

A resource-based wildlife habitat model

We define habitat quality for any particular time period as the sum of weighted resource items. More formally, habitat quality is defined for an individual time period as,

\[ HQ_{ijk} = \sum_{m=1}^{n} (R_m \times w_m), \]

where habitat quality for pixel *i* of ecosystem *j* during season *k* (*HQ*_ijk*) is estimated as the sum of the product between resource abundance *R* for the *m*-th resource item and a seasonal importance weight *w* for the same resource item. Resources would represent potential food items and be measured as the abundance of that item within a study pixel (presence, density, biomass, etc.), while importance weights would represent the seasonal significance of the food item, such as nutritional value (e.g., digestibility, energy, diet contribution, etc.). Since the timing of resource use, overall importance within a season and even identity of possible resources may vary by ecosystem, separate matrices of food items and importance weights may be necessary. In the simplest situation involving a single ecosystem, requirements would include a single list of food items, their predicted distribution (resource patches) and/or abundance, and a matrix of seasonal importance weights by resource item. To estimate an annual index of habitat quality, seasonal habitat quality values would be summed or more formally estimated as,

\[ HQ_{ij} = \sum_{k=1}^{n} \sum_{m=1}^{n} (R_{km} \times w_{km}), \]

where habitat quality value for pixel *i* of ecosystem *j* (*HQ*_ij*) is estimated as the sum of the product between resource abundance *R* in pixel *i* of ecosystem *j* for the *m*-th resource item.
item during the $k$-th season and an importance weight $w$ assigned to pixel $i$ of ecosystem $j$ for the same $m$-th resource item and $k$-th season.

Application of habitat model for grizzly bears in west-central Alberta, Canada

We applied our habitat-modeling approach to a population of grizzly bears in west-central Alberta, Canada. Resource abundance ($R$) was defined by the presence-absence (1 or 0) of critical resource (food) items ($m$) in pixel $i$ of our study area based on a diet assessment for the population from Munro et al. (2006). Importance weights ($w$) for each resource item in any one of 10 bi-monthly seasons ($k$) starting the first two weeks of May and ending on the last two weeks of September were based on percent digestible matter reported in Munro et al. 2006. Because diets varied among mountain and foothills ecosystems ($j$), we used natural region boundaries (Natural Regions Committee 2006) to define study pixels as either Rocky Mountains or Foothills (Figure 1). Although 40 total resource items exceeding 1% digestible matter were identified by Munro et al. (2006), only those items containing at least 5% digestible matter for any particular bi-monthly period and ecosystem were considered resulting in a total of 10 resource items (Table 1).

Diet-based resources fell into one of four general feeding activities: root digging, grazing-herbivory, carnivory, scavenging, myrmecophagy for animal (including insects) matter, and frugivory (Table 1). Specifically, root digging was dominated by excavation of sweet vetch ($Hedysarum$ spp.) roots during spring and late fall periods. Grazing/herbivory, on the other hand, dominated the diets of bears in late spring through mid-summer, but was well represented in all seasons. Food items included horsetails ($Equisetum$ spp.), cow-parsnip ($Heracleum lanatum$), clover ($Trifolium$ spp.), graminoids, and other forbs. In contrast to the use of root and herbaceous materials, diets represented by animal matter included scavenging/carnivorous activities of ungulates, predominately moose ($Alces alces$) in the late spring and early summer and to a lesser extent Hymenoptera insects (mostly ants, hence myrmecophagy) during mid-summer. Frugivory activities dominated the late summer diet of grizzly bears, and were represented most by Canada buffaloberry ($Shepherdia canadensis$) and black huckleberry ($Vaccinium membranaceum$).

Modeling the distribution of resource patches

Resource patches (presence-absence) for nine of 10 resource items were predicted for the study area using 642 stratified (by landcover) random field plots, a suite of environmental GIS predictors (Table 2), and binomial regression. All statistical analyses were performed in STATA 9 (Stata Corp., College Station, Texas). In total, 296 field plots occurred in un-harvested forests, 247 in clear-cut harvested forests (ages ranging up to 49 years), and 99 plots in non-forested vegetated openings, such as alpine meadows (details of field methods can be found in Nielsen et al. 2004). Presence-absence models for each resource item were estimated separately by the three habitat strata above to allow for landcover-specific responses and variables. Environmental predictors (Table 2) included both direct and indirect resources including landcover type from McDermid et al. (2005), climatic variables hypothesized to relate to critical limiting factors or resource gradients (e.g., growing degree days, frost free period, mean annual temperature, mean annual precipitation, and summer or annual moisture index), terrain-influenced environmental
conditions (compound topographic index, topographic position, and solar radiation) derived from a 30-m digital elevation model, and forest stand characteristics (age, canopy, leaf area index, and distance to edge) from McDermid et al. (2005) and forest stand inventory maps.

In contrast to plant- and insect-based resources that could be modeled from plot data, ungulate resource patches were estimated using 51 known (field-visited GPS telemetry sites) ungulate carcass (or kill) locations used by grizzly bears and reported in Munro et al. (2006). Ungulate use locations were compared with 5100 random available locations generated with the Generate Random Points tool in Hawth’s Analysis Tools for ArcGIS (Beyer 2004). A 99% fixed kernel polygon (2139-km²) estimated from field-visited GPS telemetry locations and the Fixed Kernel Density Estimator tool in Hawth’s Analysis Tools for ArcGIS (Beyer 2007) was used to constrain available locations to areas where animal use was recorded. Because habitat conditions suitable for ungulates necessitated examination of habitat at multiple scales, we examined environmental predictors for three possible scales ranging from local site conditions (30-m pixel) to a moose home range (Lynch 1986) scale of 51.6-km² (4053-m radius moving window). An intermediate scale of 51.5 ha (405-m radius moving window) was also considered. Moose home ranges were chosen since moose were most frequently preyed upon or scavenged by grizzly bears in the region (Munro et al. 2006). Because absent locations were unknown, a resource selection function (Manly et al. 2002) model was used to estimate the distribution of ungulate resources. Ratio of use and available locations were balanced by weighting available locations at 0.01, thus resulting in a sample size of 51 carcass locations and 51 random locations (102 total samples).

Model-building procedures for all resource items followed that of Hosmer & Lemeshow (2000) with univariate analyses used to rank the importance of individual linear factors or hypothesized non-linear forms and interaction terms. The highest ranked uncorrelated (|r| < 0.7) factors were introduced individually in a forward manner until all significant (p < 0.1) factors were retained. Once multivariate models were selected for each resource item, model significance was assessed using a likelihood ratio $\chi^2$ test. Model accuracy was evaluated using receiver operating characteristics (ROC) area under the curve (AUC) with AUC values of 0.9 and above representing high model accuracy, 0.7 to 0.9 good model accuracy, and <0.7 low model accuracy (Swets 1988; Manel et al. 2001). To classify presence-absence (resource patches) for each resource item from probability of occurrence models (by each broad habitat strata), the minimum absolute difference between sensitivity and specificity values were used to estimate an optimal cutoff probability (Liu et al. 2005) with probabilities greater than or equal to the cutoff probability predicted as present and less than the cutoff probability as absent.

Resource patches (30-m pixel grain size) for all 10 items were mapped in a GIS using model coefficients, associated GIS predictors, and the raster calculator function in the spatial analyst extension of ArcGIS 9.1 (ESRI, Redlands, California). Mapped probabilities were reclassified to binary presence-absence (1 or 0) maps (resource patches) using the cut-off probability estimated from the sensitivity-specificity analysis and the reclassify function in ArcGIS spatial analyst.
Temporal importance (availability) of resource patches
Importance weights for each resource item (detailed in Appendix I) were assigned to each of the 10 pre-defined seasons (bi-monthly periods) based on percent digestible matter reported in Munro et al. (2006). In effect, the seasonal weighting of resource patches resulted in 100 separate spatial models (10 resources X 10 seasons). To estimate potential habitat quality, weighted resource patches were summed in a GIS for each bi-monthly period using the ArcGIS raster calculator function of spatial analyst. Annual (multi-seasonal) potential habitat quality was estimated from the sum of seasonal habitat quality models.

Resource patches and animal foraging locations
We used 1032 field-visited animal use (GPS telemetry) locations from nine sub-adult and adult female grizzly bears in west-central Alberta (Munro et al. 2006) to evaluate the use of resource patches. At each of 1032 animal use locations, activity was classified as bedding, foraging (by type or food item), or no obvious sign. Using locations where foraging activity was recorded for sweet vetch, cow-parsnip, Hymenoptera insects, buffaloberry, and ungulates (kill sites), predicted probability of occurrence for each of the five resource items were recorded as well as at 2139 random locations (sample intensity of 1 location per km²) estimated from the Generate Random Points tool in Hawth’s Analysis Tools for ArcGIS (Beyer 2007) within the 99% fixed kernel polygon of field visited use locations estimated for the ungulate carcass model. Binomial regression was used to compare probability of occurrence and classified presence-absence at use locations (1s) compared with random locations (0s). Although these data represent an independent assessment for plant-based resources, evaluations are likely liberal for ungulate resources since the same locations were used for model building. Odds ratio and significance of grizzly bears use of predicted resource patches compared to random locations are reported.

Potential habitat quality and animal use (telemetry) locations
In addition to assessing use of individual resource patches, we also used 42853 animal use locations from 44 GPS radiocollared grizzly bears to assess selection of potential habitat quality models. Although not all telemetry locations would represent foraging activities, a positive relationship between potential habitat quality and animal use was hypothesized. To specifically test whether potential habitat quality was positively related to observed animal use, seasonal habitat quality at telemetry locations were compared to random available habitat quality for each animal using seasonal (bi-monthly) RSFs (Manly et al. 2002). Habitat availability was defined for an animal’s multi-annual minimum convex polygon (MCP) home range (design III, Thomas & Taylor 1990) with a sampling intensity of 1 location per 1-km² using the Generate Random Points tool from Hawth’s Analysis Tools for ArcGIS (Beyer 2007). Resource availability was assumed to be constant among seasons to ensure that seasonal changes in selection of habitats were due to habitat use and not the method of available sampling. Given animal-specific estimates of selection for each of the 10 seasons, population-level seasonal selection (expressed as odds ratios) was estimated as the mean of individual animal selection following a two-stage modeling process (Cox & Hinkley 1974 in Manly et al. 2002). Population-level significance was estimated using a one-sample t-test ($H_0$: mean $> 1$).
Realized habitat and regional patterns of grizzly bear occupancy-abundance

We used data from a DNA mark-recapture hair-snag (Woods et al. 1999) study (8820-km² area of foothills) reported in Boulanger et al. (2005) to estimate regional patterns of occupancy (detected) and relative abundance (number of detected individuals) of grizzly bears. In contrast to telemetry locations that occur only within occupied range, hair-snag sample locations (707 sites) occurred in both occupied to unoccupied (undetected) grizzly bear range necessitating a modification to our potential habitat quality model to reflect historic displacement of grizzly bears. We estimated realized habitat quality by down-weighting potential habitat quality using a model of regional female grizzly bear occupancy that was based on all known female grizzly bear minimum convex polygon home ranges south of the city of Grande Prairie, Alberta. A systematic grid of points located every 10-km was generated for the region in a GIS using the Generate Regular Points tools from the Hawth’s Analysis Tools for ArcGIS (Beyer 2007). All systematic locations were classified as occupied (occurring within a known home range) or unoccupied (occurring outside a known home range). Proportion agriculture and natural sub-region categories within a 10-km radius were used as predictors of regional female grizzly bear occupancy with binomial regression used to estimate the probability of occupancy. Realized habitat quality was estimated as the product of potential habitat quality and probability of regional female grizzly bear occupancy.

Using the model of realized habitat quality, we evaluated whether habitat conditions were significantly related to patterns of grizzly bear occupancy and abundance measured at hair-snag locations. We used a zero-inflated count (Poisson) model (Barry & Welsh 2002) with realized habitat quality as predictors of occurrence and abundance (number of detected individuals). The zero-inflated count models assumed two latent groups: group A, the inflation part that only represented absent locations; and group ~A, which represented the counts including a possible zero count (Long & Freese 2003). Although zero-inflation was likely required given the preponderance of absences (642 of 707 hair-snag sites where categorized as undetected), we ensured that a zero-inflated count model was necessary through a Vuong (V) test (Vuong 1989). Because occupancy and abundance are unlikely to be explained from very local habitat conditions, realized habitat quality around hair-snag locations were examined for four possible scales: (1) 300-m radius or 10 times the original map grain; (2) 1.69-km radius or detection scale of hair-snag sites from Boulanger et al. 2004; (3) 3.95-km radius or the scale of the DNA sampling grid used for locating ‘re-capture’ sites; and (4) 10-km radius or approximately the 90% kernel home range of female grizzly bears in the area (Nielsen et al. 2008). All scales were evaluated, with the most significant scale reported.

Results

Distribution of resource patches

Likelihood ratio model $\chi^2$ statistics and ROC AUC results indicated that all resource patch models were significant and all but forbs and clover in harvested forests had ‘good’ to ‘high’ predictive accuracy (Tables 3-6). Lower predictive accuracy (AUC <0.7) for forbs and clover in harvested forests suggest that other factors, such as site preparation, may be more important for determining local occupancy within a clear cut than environmental factors. Non-linear relationships in species occurrence were common for many environmental factors demonstrating the presence of broad environmental gradients
over a relatively small region. Below, we describe environmental relationships and spatial patterns by each resource group (e.g., roots, herbaceous matter, animal matter, and fruit).

Factors affecting the distribution of rooting resources
Rooting resources, represented by sweet vetch, were most common to open conifer and treed bog landcover types when compared to closed conifer forests (Table 3). Forest stands with high soil wetness or canopy closure (negative interaction), short frost free periods, and low and high growing degree days (base 0° C) were positively related to sites with sweet vetch occupancy (Table 3). In clear-cut forests, sweet vetch occupancy increased in sites with higher soil wetness and lower landscape position, areas of low growing season precipitation, and areas of low or high growing degree days at a base of 0° C (Table 4). In non-forested stands, sweet vetch occurrence was lower in anthropogenic habitats compared to herbaceous habitats (Table 5). Mean annual temperature and changes in early to late summer leaf area, as well as areas of low or high soil wetness, were positively related to sweet vetch occurrence for non-forested sites (Table 5). Maps predicting the distribution of sweet vetch illustrated that the species was most common to mountain environments, but also scattered in riparian habitats and low elevation sites in the foothills (Figure 2a).

Factors affecting the distribution of herbaceous resources
Herbaceous resources, represented by horsetails, forbs, graminoids, cow-parsnip, and clover, were varied in distribution. Below we describe environmental relationships for each resource item.

Horsetails were more common in mixed forest and especially treed bog sites, and less common in open conifer and deciduous forests when compared with closed conifer stands (Table 3). Horsetail occupancy was highest in areas of moderate mean annual temperature, while also increasing in forest stands with high soil wetness and canopy with a negative interaction between canopy and soil wetness reducing horsetail occupancy in wet areas of high canopy cover (Table 3). Horsetail occurrence in clear-cut forests was highest in older clear-cuts located in lower landscape positions (Table 4). In non-forested sites, horsetail occurred more frequently in shrub, anthropogenic and especially open bog habitat when compared with herbaceous meadows (Table 5). Both soil wetness and growing degree days at base 5° C positively influenced horsetail occurrence in non-forested habitats (Table 5).

Forb resources, other than that of cow-parsnip and clover, were more common in mixed forests, open conifer forests, treed bog stands, and deciduous forests when compared with closed conifer forests (Table 3). Forb occupancy in forest stands was highest in young forests with low annual moisture and either low or high amounts of leaf area (Table 3). In clear-cut forests, forb occupancy was highest in older clear-cuts with short or long frost free periods and higher growing degree days at a base 0° C (Table 4). Forbs were ubiquitous to non-forested sites.

Graminoids were less common in treed bog and more common in deciduous forests, open conifer, and mixed forests when compared with closed conifer stands (Table 3).
Graminoid occupancy was highest in forest stands having low growing season precipitation, low canopy cover, low or high levels of leaf area during early summer, and in areas with low or high solar radiation (Table 3). In clear-cut forests, graminoid occurrence was highest in areas with moderate annual precipitation and either low or high soil wetness (Table 4). Finally, in non-forested sites graminoids were positively related to late summer leaf area index, less common to shrub and open bog habitats when compared to herbaceous habitats, and found in all anthropogenic sites (Table 5).

Cow-parsnip was less common in mixed forests, deciduous forests, and especially open conifer forests when compared with closed conifer stands, with no cow-parsnip detected in treed bog habitats (Table 3). Cow-parsnip occupancy increased in forest stands having a moderate frost free period, a high late-summer leaf area index, and low or high amounts of solar radiation and soil wetness (Table 3). In contrast to unharvested forests, cow-parsnip occurrence in clear-cuts increased with increasing solar radiation and topographic position (Table 4). No cow-parsnip was detected in non-forested habitats.

Clover was more common in treed bog, mixed forest, deciduous forests, and especially open conifer stands when compared to closed conifer stands (Table 3). Clover occupancy was highest in forest stands having a moderate frost free period and in sites that were close to open edges (Table 3). In clear-cut forests, clover occupancy peaked in sites that experienced large-scale changes in leaf area index between early and late summer, areas of high soil wetness, and in areas with either low or high mean annual precipitation (Table 4). In non-forested habitats, clover was undetected in open bog and shrub habitats, while being similar for anthropogenic habitats and herbaceous openings. Clover occupancy was highest in non-forested habitats that had moderate levels of summer moisture (Table 5).

Spatial predictions of herbaceous resources illustrate the wide-spread availability—excluding barren mountains—of herbaceous matter (Figure 2b).

Factors affecting the distribution of animal resources
Hymenoptera insects were more common in mixed forest, deciduous forest, treed bog, and especially open conifer forests when compared with closed conifer stands (Table 3). In forested stands, Hymenoptera occurrence was negatively related to stand age, canopy cover, and soil wetness, although occupancy did increase in wet sites having high canopy coverage (i.e., a positive interaction) and in forests having low or high solar radiation (Table 3). In clear-cut forests, Hymenoptera occurrence was negatively affected by frost free period, while being positively associated with intermediate aged stands and areas of moderate late summer leaf area (Table 4). In non-forested sites, Hymenoptera insect occupancy was highest in herbaceous sites compared to open bog, anthropogenic habitats, and especially shrub habitats (Table 5). Hymenoptera were also more common in non-forested habitats when near forest edges, in areas of moderate summer moisture, low or high late summer leaf area, low or high solar radiation, and areas of higher topographic position (Table 5). Spatial predictions of Hymenoptera insects illustrated that they were most prevalent in the Foothills (Figure 2c).
Unlike plant and insect-based resources where local site conditions were used to describe resource patches, habitats associated with ungulate carcass sites (kill locations) were best described by shrub and clear-cut forest composition at scales representing moose home ranges (i.e., 51.6 km²), canopy conditions at the site (30 m pixel), and local patterns (51.5 ha) of soil wetness and landscape position (Table 6). Areas of intermediate canopy, low topographic position, high soil wetness, moderate levels of clear-cut harvesting, and high composition of shrub habitat best described areas of ungulate habitat where carcass or kill sites were most common (Table 6). Spatial predictions of ungulate resources illustrated that their distribution was patchy, but generally highest in the foothills where shrub and regenerating clear-cut forests are most common (Figure 2c).

Factors affecting the distribution of fruiting resources
Buffaloberry was less common in treed bog and more common in open conifer, deciduous forest, and mixed forest stands when compared with closed conifer forests (Table 3). Buffaloberry occurrence in forested stands was positively related to areas of high solar radiation, low topographic position, moderate growing season precipitation, and moderate forest age (Table 3). In clear-cut forests, buffaloberry occurrence was positively related to annual moisture, early summer leaf area, and areas of short or long frost free periods (Table 4). In non-forested habitats, buffaloberry occurrence was higher in anthropogenic habitats when compared to herbaceous habitats, with growing degree days (base 0°C) and solar radiation negatively affecting buffaloberry occupancy and either low or high late summer leaf area (Table 5). Spatial predictions of buffaloberry illustrate the wide-spread distribution of the species, especially at lower elevations (Figure 2d).

In contrast to buffaloberry, black huckleberry was most common in closed conifer stands with decreasing occurrence in mixed forest, open conifer, and deciduous forests (Table 3). Black huckleberry was not detected in treed bog habitats. Black huckleberry occupancy in forested stands was positively related to growing season precipitation, negatively related to changes in leaf area index among early and late summer periods, and in areas with moderate late summer leaf area (Table 3). In clear-cut forests, black huckleberry was negatively related to solar radiation with occupancy peaking in areas of moderate early summer leaf area and low or high annual moisture (Table 4). Black huckleberry was absent (undetected) from non-forested sites. Spatial predictions of black huckleberry suggest it to be common throughout forested environments in the region (Figure 2d).

Resource patches and animal foraging locations
Resource patches of sweet vetch, cow-parsnip, Hymenoptera, and buffaloberry were all used more than random at an odds of 2.4, 1.9, 2.1, 5.2, and 8.7 times respectively (Table 7). All models were significant ($p<0.1$) except for cow-parsnip (marginally non-significant), which likely reflects the small number of documented cow-parsnip feeding locations ($n=25$). Using unclassified models of probability of occurrence, all models were significant, including cow-parsnip, with odds ratios ranging from a low of 7.0 (SE = 1.8, $p<0.001$) for Hymenoptera insects to a high of 68.9 (SE = 41.8, $p<0.001$) for ungulate carcass sites (Table 7). Higher odds and greater significance for probability of
occurrence models indicate a loss of information during classification of resource occupancy. However, given positive relationships with resource patch models, such losses may not be consequential.

**Potential habitat quality and animal use (telemetry) locations**

Potential habitat quality varied spatially and temporally (bi-monthly) reflecting widespread differences in the distribution of food resources and temporal changes in grizzly bear diets (Figure 3a, 3b, & 3c). Considering all bi-monthly periods together (multi-seasonal habitat quality), habitat values were highest in low elevation mountain valleys and young forest stands in the Upper Foothills (Figure 3d). Despite high values in mountain valleys, much of the Rocky Mountain region was non-vegetated (38.3% of Alpine/Sub-Alpine and 11.6% of Montane versus 3% for the Foothills) reducing overall habitat conditions for natural region. Although some areas contained lower multi-seasonal habitat, variability was less pronounced. Few areas contained all possible resource items reducing multi-seasonal habitat quality well below the possible maximum (Figure 3d).

Seasonal telemetry locations of grizzly bears were positively related to potential habitat quality, although strength of relationships were highest during the mid-summer to late-summer periods (late July and early August) when diets shifted from herbivory to frugivory (Figure 4). Odds (per standard deviation increase in habitat quality) of selecting high-quality habitats compared with random home range locations ranged from a low of 1.3 in early May and early September to 1.6 during early August (Figure 4). Assuming a three-standard-deviation increase in habitat quality, bears would be predicted to increase use by a factor of 4 to 5. No significant differences among sex were found. Reporting potential habitat quality by landcover type, open conifer had the highest score at 53.5, while treed bog had the lowest potential score at 19.3 (Figure 6).

**Realized habitat and regional patterns of grizzly bear occupancy-abundance**

Regional female grizzly bear occupancy (range) was negatively related to the proportion of agriculture within 10 km, and positively related to alpine, sub-alpine, montane, and Upper Foothills natural sub-regions when compared with a reference (indicator) category of Lower Foothills (Table 8). Occupancy in the Central Mixedwood natural sub-region was significantly lower than the Lower Foothills, while occupancy in Foothills Parkland and Foothills Fescue was similar to the Lower Foothills (Table 8). Given these patterns, realized habitat quality was highest in sub-alpine valleys and lowest in the Lower Foothills (Figure 5a). Considering displacement of historic grizzly bear populations from resource-rich habitats, estimated habitat deficit (absolute habitat loss) was highest in the Montane region of the Rocky Mountains (Figure 5b). Specific to landcover types, open conifer had the highest realized habitat, while treed bog had the lowest realized habitat (Figure 6). Habitat deficits were most pronounced for landcover types common to the Montane and Lower Foothills region. In particular, deciduous and mixed forests demonstrated the most substantial losses, while closed conifer was the most stable landcover type (Figure 6).
A zero-inflated Poisson model describing the relationship between realized habitat quality and occupancy-abundance at DNA hair-snag sites was significant overall ($\chi^2 = 7.84, p = 0.005$) with local scales (300-m) best describing occupancy and regional scales (10-km) best describing relative abundance of grizzly bears (Table 9). A significant Voung test ($V = 6.02, p<0.001$) confirmed the need for a zero-inflated process. The model predicted that for each standard deviation increase in realized habitat quality, there would be an 81% (±13%) increase in occupancy and a 32% (±12%) increase in grizzly bear abundance (Table 9).

**Discussion**

Grizzly bears are especially well-suited to food-based definitions of habitat, since they are a generalist species characterized by a diverse and omnivorous diet that affects body condition (Pritchard & Robbins 1990; Hilderbrand *et al.* 1999a; Rode *et al.* 2001) and population density (Hilderbrand *et al.* 1999b). A diverse diet also provides plasticity in habitat requirements (Pasitschniak-Arts 1993) making definitions of habitats using telemetry locations and indirect environmental factors difficult. Certainly, habitat-selection studies have enlightened local knowledge of habitat use, while also providing a wealth of information to generate hypotheses for further study. Rarely, however, are there any evaluations of whether habitat selection relates to population-level measures such as abundance or density (Boyce & McDonald 1999). Nielsen *et al.* (2005) and Johnson *et al.* (2008) suggest that habitat selection estimates should not be assumed to scale with local density or regional predictions of abundance.

With this work, we describe an alternative approach for measuring and mapping habitat quality based on distribution models for critical food items (resource patches) and knowledge of seasonal diets (i.e., timing of resource use). We illustrated our modeling approach for a population of grizzly bears in west-central Alberta. Philosophically, our approach and definition of habitat is similar to grizzly bear habitat effectiveness models (Mattson & Knight 1991; Gibeau 1998) used for cumulative effects analyses and coefficients of habitat productivity by Mattson *et al.* (2004). Mattson *et al.* (2004) indexed nutrient and energetic levels of food resources to scale habitat productivity by landcover type, while we simply used percent digestible matter from a diet study (Munro *et al.* 2006) to scale habitat conditions to species distribution (resource patch) models. Using our model of grizzly bear habitat, we demonstrated a significant, positive relationship between habitat quality and regional patterns in grizzly bear occupancy-abundance and local patterns of seasonal habitat use. Considering food resource distribution and regional diet, open conifer forests were predicted to be the most important habitat type. Regionally, montane areas of the Rocky Mountain natural region contained the highest habitat potential, but also exhibited the greatest habitat loss (habitat deficit) due to human activities and historic patterns of mortality. We suggest that maps of habitat deficit be considered when prioritizing conservation actions for population recovery.

**Diet-based versus telemetry-based habitat models**

Diet-based wildlife habitat modeling has a number of advantages over traditional telemetry-based wildlife habitat modeling techniques such as resource selection functions.
(RSFs; Manly et al. 2002). First, if diets are known, resource-based models can be estimated without having a sample of animal locations. This avoids complications relating to definitions of random available samples and issues of sample contamination inherent in RSFs (Keating & Cherry 2004). Animal capture and collaring is also avoided, which can be difficult or cost prohibitive for some species and environments and may negatively affect the population (Cattet et al. 2008). Although animal locations are unnecessary, information on the distribution of food resource items are needed. For herbivores and for the most part omnivores, distribution and abundance of resource items can be easily estimated using standard vegetation sampling (existing plots or new surveys). We used vegetation plots to estimate the distribution for 9 of 10 critical food items for a population of grizzly bears (omnivore) and found excellent fit between predicted resource patches and habitat use.

Because many populations are limited by resource abundance (Miyashita 1992; Mattson et al. 2004; Brasher et al. 2007), a second advantage to diet-based habitat models is that they explicitly target critical limiting factors (resources). Habitat units are intuitive and meaningful, easily allowing for multi-seasonal estimates (sums) of habitat quality. In comparison, it is often difficult to interpret and scale telemetry-based wildlife habitat modeling, especially when based on indirect variables (Austin 2007). RSF-based habitat maps, for instance, are typically binned into ordinal categories of relative probability of habitat use (Boyce et al. 2002). Interpretation is complicated by the lack of actual probabilities (Lele & Keim 2006), while estimation of multi-seasonal habitat from summed seasonal maps assume that bins (ordinal groups) are constant in number and classification. Our diet-based method allows flexible, yet meaningful definitions of multi-seasonal habitat, since units are consistent. In our model we described habitats in units of available percent digestible matter.

A third advantage of diet-based habitat models is the fact that temporal dynamics (phenology) in resource items are explicitly considered. Many animals (especially generalists) have evolved an adaptive switching strategy (Ma & Levin 2006) where habitat use changes based on availability of resources (Schwemmer & Garthe 2008). Diet-based approaches explicitly identify changes in resource availability, while telemetry-based models often combine seasons to boost statistical power or examine patterns in habitat use to define seasons in a post hoc manner without identifying or acknowledging the mechanisms responsible for habitat switching. Management and conservation habitats are compromised when mechanisms are unknown.

A fourth advantage of diet-based models is that animal activities, such as bedding and movement, are not emphasized as critical habitat like they typically are in telemetry-based habitat models. For many species, including grizzly bears, bedding locations are not limiting. Although movement or corridor habitat can be limiting (Chetkiewicz et al. 2006), connectivity is frequently best considered at the scale of populations (Urban & Keitt 2001), not within an animal’s home range – a frequent scale of telemetry-based habitat modeling. In fact, examining connectivity of resource patches with or without human disturbance may be more appropriate than using a general habitat selection map that uses all telemetry locations.
Finally, diet-based habitat models provide a measure of habitat potential. For wildlife populations that have declined or have been regionally displaced due human activities, such as hunting or poaching, habitat potential can be quantified and when area of displacement is known, an index of habitat loss or habitat deficits can be estimated. Such information can be used to select or prioritize sites for restoration or reintroduction or to compare restoration strategies. Restoration potential and habitat loss are difficult to quantify with telemetry-based habitat models, since they only measure realized habitat conditions.

Areas of future model improvement
Annual variations in resource abundance should be considered in wildlife habitat models, since resource abundance can be highly variable among years (Suzuki et al. 1998; Brown & Sherry 2006; Marciniak et al. 2007). Fruit and nut production, in particular, exhibits supra-annual variations (Abrahamson & Layne 2003; Layne & Abrahamson 2004; Brearley et al. 2007). This pulsing of resources influences body condition and population dynamics for many wildlife species (McShea 2000; Wong et al. 2005) and should be accounted for. Pelchat & Ruff (1986), for example, demonstrated that when fruit production of velvet-leaved blueberry (Vaccinium myrtilloides) averaged 423 berries per square metre, black bears (Ursus americanus) gained weight, while black bears lost weight when fruit production averaged 66 berries per square metre. Variations in resource abundance also affect populations. For instance, in years of resource scarcity, bear-human conflicts and mortality rates of grizzly bears increased in Yellowstone National Park (Mattson et al. 1992; Gunther et al. 2004). Likewise, hunting success of black bears in Minnesota (as well as proportion females harvested) increased when hazelnut (Corylus spp.) and acorn (Quercus spp.) abundance was low (Noyce & Garshelis 1997).

When annual differences in fruit or nut production are known, additional parameters can be added to habitat models to scale the annual importance of resources. For example, Mattson et al. (2004) classified years of good and poor whitebark pine (Pinus albicaulis) seed crop consumption by grizzly bears when scaling habitat values. Although varying habitat conditions annually is especially important for demographic studies, such actions would limit their application in scenario (habitat projection) assessments (i.e., Nielsen et al. 2008) or conservation mapping exercises (i.e., Nielsen et al. 2006) unless trends or cycles in resources are known, inter-annual generalizations are made, or random sequences of years selected for projections. From a conservation perspective, perhaps the most important objective is simply identifying (mapping) critical resource patches, which does not necessitate annual scaling of habitat models. Regardless, wildlife habitat modeling should consider annual variations in habitat conditions for species that use highly variable resources.

Although not restricted to our general habitat model, we used resource patches defined by the presence-absence of resource items to measure resource availability. Actual abundance (density or grams of resource), or better yet resource nutrition (e.g., starch, protein, carbohydrate, etc.) and energetics (e.g., kilocalories or potential net digestible
energy) should be considered. Mattson et al. (2004) based their grizzly bear habitat model on potential net digestible energy, although values were restricted to landcover types, rather than individual species distribution models, which limit their application.

**Conclusions**

Using a simple resource-based habitat model, distribution models of resources, and knowledge of seasonal diets, we successfully predicted regional patterns in grizzly bear occupancy-abundance and local seasonal patterns of habitat use. Spatial predictions of critical grizzly bear resources and diet-based seasonal weighting of resource patches provided a mechanistic alternative to telemetry-based habitat selection modeling that relies on indirect resource variables and suffers from a number of technical problems relating to model development and application. Although our models of habitat quality predicted the distribution and abundance of grizzly bears, resource patches were defined by the presence-absence of food items. Measures of resource abundance/productivity (e.g., grams of available food), or energetic characteristics (e.g., kilocalories or potential net digestible energy) should be considered, as well as annual variations in those resources. We are currently evaluating the potential to include energetics, as well as expanding model products from the core Hinton study to other regions, including the northern population where RSF models are lacking.

**References**


Table 1. Critical food resources used to define habitat in west-central Alberta, Canada. Food resource, abbreviation (code), feeding activity, and general season of use are described.

<table>
<thead>
<tr>
<th>Food resource</th>
<th>Code</th>
<th>Feeding activity</th>
<th>Season of use</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Equisetum</em> spp. (horsetail)</td>
<td>EQAR</td>
<td>grazing-herbivory</td>
<td>spring</td>
</tr>
<tr>
<td>Forbs</td>
<td>FORB</td>
<td>grazing-herbivory</td>
<td>spring – summer</td>
</tr>
<tr>
<td>graminoids</td>
<td>GRAS</td>
<td>grazing-herbivory</td>
<td>spring – summer</td>
</tr>
<tr>
<td><em>Hedysarum</em> spp. (sweet vetch)</td>
<td>HEDY</td>
<td>root digging</td>
<td>spring and fall</td>
</tr>
<tr>
<td><em>Heracleum lanatum</em> (cow parsnip)</td>
<td>HELA</td>
<td>grazing-herbivory</td>
<td>summer</td>
</tr>
<tr>
<td>Hymenoptera insects (mostly ants)</td>
<td>HYME</td>
<td>myrmecophagy</td>
<td>summer</td>
</tr>
<tr>
<td><em>Shepherdia canadensis</em> (buffaloberry)</td>
<td>SHCA</td>
<td>frugivory</td>
<td>late summer – fall</td>
</tr>
<tr>
<td><em>Trifolium</em> spp. (clover)</td>
<td>TRRE</td>
<td>grazing-herbivory</td>
<td>spring – summer</td>
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<tr>
<td>ungulate carcass (scavenging or kill site)</td>
<td>UNGL</td>
<td>carnivory/scavenging</td>
<td>spring – early summer</td>
</tr>
<tr>
<td><em>Vaccinium membranaceum</em> (huckleberry)</td>
<td>VAME</td>
<td>frugivory</td>
<td>late summer and fall</td>
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Table 2. Description and characteristics of environmental variables used to model the probability of occurrence of individual grizzly bear food resources in west-central Alberta, Canada.

<table>
<thead>
<tr>
<th>Variable group</th>
<th>Variable name</th>
<th>Abbrev.</th>
<th>Res.</th>
<th>Units</th>
<th>Data range</th>
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<td>category</td>
<td>0 or 1</td>
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<td>30 m</td>
<td>category</td>
<td>0 or 1</td>
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<td>30 m</td>
<td>category</td>
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<td>category</td>
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<td>degree days</td>
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<td>DD5</td>
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<td>degree days</td>
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<td></td>
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<td>FFP</td>
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<td>days</td>
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<td>percent</td>
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<td></td>
<td>forest age</td>
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<td>30 m</td>
<td>10 years</td>
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<td>leaf area index (10-Jun; 161&lt;sup&gt;st&lt;/sup&gt; day)</td>
<td>LAI-163</td>
<td>30 m</td>
<td>cm&lt;sup&gt;2&lt;/sup&gt;/m&lt;sup&gt;2&lt;/sup&gt;</td>
<td>0.62–6.01</td>
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<td>LAI-223</td>
<td>30 m</td>
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<td>% change in LAI (161 to 225 day)</td>
<td>ΔLAI</td>
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<td>percent</td>
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<td>30 m</td>
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Table 3. Estimated coefficients describing the occurrence of 9 grizzly bear food resources (see Table 1 for definitions of 4 letter codes) in non-harvested forest stands near Hinton, Alberta.

<table>
<thead>
<tr>
<th>Variables</th>
<th>Roots</th>
<th>Herbaceous matter</th>
<th>Animal</th>
<th>Fruit</th>
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<td>HEDY</td>
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<td>FORB</td>
<td>GRAS</td>
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<tr>
<td><strong>Habitat type</strong></td>
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<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>DFOR</td>
<td>-0.245</td>
<td>-0.661</td>
<td>2.09</td>
<td>0.712</td>
</tr>
<tr>
<td>MFOR</td>
<td>0.156</td>
<td>0.355</td>
<td>1.04</td>
<td>1.83</td>
</tr>
<tr>
<td>OCON</td>
<td>1.36</td>
<td>-0.149</td>
<td>1.05</td>
<td>0.881</td>
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<tr>
<td>TBOG</td>
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<td>1.42</td>
<td>-1.23</td>
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<tr>
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<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>AMI</td>
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<td></td>
</tr>
<tr>
<td>DD0</td>
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<td></td>
</tr>
<tr>
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<td>FFP²</td>
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<td>-0.016</td>
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<tr>
<td>GSP</td>
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<tr>
<td>GSP²</td>
<td></td>
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<tr>
<td>MAT</td>
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<td></td>
<td></td>
</tr>
<tr>
<td>MAT²</td>
<td></td>
<td>-0.297</td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Stand conditions</strong></td>
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<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>AGE</td>
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<td>-0.094</td>
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<tr>
<td>AGE²</td>
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<td>-1.123</td>
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<tr>
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<tr>
<td>ΔLAI</td>
<td></td>
<td></td>
<td></td>
<td>-0.033</td>
</tr>
<tr>
<td><strong>Terrain</strong></td>
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<tr>
<td>CTI</td>
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<td>0.552</td>
<td>-1.12</td>
<td>-0.323</td>
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<td>CTI²</td>
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<td>0.055</td>
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<tr>
<td>SOLR</td>
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<td>-3.74</td>
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<td>SOLR²</td>
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<td>0.024</td>
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<tr>
<td>TOPO</td>
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<td>-0.005</td>
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<tr>
<td><strong>Interactions</strong></td>
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<td>CTI x canopy</td>
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<td>-0.093</td>
<td></td>
<td>0.052</td>
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<td><strong>Model evaluation</strong></td>
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<td></td>
<td></td>
</tr>
<tr>
<td>LR²</td>
<td>78.1*</td>
<td>54.7*</td>
<td>39.1*</td>
<td>60.8*</td>
</tr>
<tr>
<td>ROC</td>
<td>0.888</td>
<td>0.73</td>
<td>0.729</td>
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<td>cutoff prob.</td>
<td>0.163</td>
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<td>0.71</td>
<td>0.788</td>
</tr>
</tbody>
</table>

§Closed conifer used as reference category in indicator contrasts of forest habitat types; † Coefficients 1000 times their original value; ‡ Coefficients 10,000 times their original value; * Significant at the p<0.01 level.
Table 4. Estimated coefficients describing the occurrence of 9 grizzly bear food resources (see Table 1 for definitions of 4 letter codes) in harvested forest stands near Hinton, Alberta.

<table>
<thead>
<tr>
<th>Variables</th>
<th>Roots</th>
<th>- - - - - - - - - - - Herbaceous matter - - - - - - -</th>
<th>Animal</th>
<th>- - Fruit - -</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>HEDY</td>
<td>EQAR FORB GRAS HELA TRRE</td>
<td>HYME SHCA VAME</td>
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<td>AMI²</td>
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<tr>
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</tr>
<tr>
<td></td>
<td>DD5</td>
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</tr>
<tr>
<td></td>
<td>DD5²</td>
<td>-0.493‡</td>
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<td>FFP</td>
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<td></td>
<td></td>
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<tr>
<td></td>
<td>FFP²</td>
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<td></td>
</tr>
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<td>GSP</td>
<td>-0.061</td>
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<tr>
<td></td>
<td>MAP</td>
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<td></td>
<td>MAP²</td>
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<td>0.415†</td>
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<tr>
<td></td>
<td>LAI-225²</td>
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<tr>
<td></td>
<td>ΔLAI</td>
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</tr>
<tr>
<td>Terrain</td>
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<tr>
<td></td>
<td>TOPO</td>
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<td>48.0*</td>
<td>19.1*</td>
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<td>cutoff prob.</td>
<td>0.108</td>
<td>0.415</td>
<td>0.699</td>
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</table>

† Coefficients 1000 times their original value; ‡ Coefficients 10 000 times their original value; * Significant at the $p<0.01$ level.
Table 5. Estimated coefficients describing the occurrence of 6 grizzly bear food resources (see Table 1 for definitions of the 4 letter codes) for herbaceous open habitats near Hinton, Alberta. Forbs occurred in every open habitat type (ubiquitous presence), while HELA and VAME where absent from all open sites.

<table>
<thead>
<tr>
<th>Variables</th>
<th>Roots</th>
<th>- - - Herbaceous matter</th>
<th>Animal</th>
<th>Fruit</th>
</tr>
</thead>
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<td></td>
<td>HEDY</td>
<td>EQAR</td>
<td>GRAS</td>
<td>TRRE</td>
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<tr>
<td>Habitat type§</td>
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<td></td>
<td></td>
</tr>
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<tr>
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</tr>
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<td>SMI</td>
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<tr>
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<tr>
<td>EDGE</td>
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<td></td>
</tr>
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<td>LAI-225</td>
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</tr>
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<tr>
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</tr>
<tr>
<td>SOLR²</td>
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</tr>
<tr>
<td>TOPO</td>
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</tr>
<tr>
<td>TOPO²</td>
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</tr>
<tr>
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<td>0.699</td>
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<td>Model evaluation</td>
<td></td>
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<td></td>
<td></td>
</tr>
<tr>
<td>LR²</td>
<td>24.0*</td>
<td>22.4*</td>
<td>14.4*</td>
<td>25.1*</td>
</tr>
<tr>
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<td>0.889</td>
<td>0.77</td>
<td>0.822</td>
<td>0.905</td>
</tr>
</tbody>
</table>
cutoff prob. | 0.211 | 0.39  | 0.79  | 0.355 | 0.405 | 0.089 |

§herbaceous openings used as reference category in indicator contrasts of habitat types; † Coefficients 1000 times their original value; ‡ Coefficients 10 000 times their original value. * Significant at the p<0.01 level.
Table 6. Ungulate carcass (kill site) model (LR $\chi^2 = 37.0, p<0.001, DF = 7$) describing the relative probability of a site being classified as an ungulate (primarily moose) kill location (optimal cut-off probability of 0.5497). Model accuracy was good (AUC = 0.823) with excellent model fit (Hosmer & Lemeshow GOF $\chi^2 = 112.63, p = 1.0$).

<table>
<thead>
<tr>
<th>Variable</th>
<th>Scale</th>
<th>Coef.</th>
<th>SE</th>
<th>$p$</th>
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</thead>
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<tr>
<td>CNPY</td>
<td>30-m pixel</td>
<td>0.097</td>
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<td>0.025</td>
</tr>
<tr>
<td>CNPY$^2$</td>
<td>30-m pixel</td>
<td>-0.010</td>
<td>0.005</td>
<td>0.034</td>
</tr>
<tr>
<td>TOPO</td>
<td>51.5 ha</td>
<td>-0.019</td>
<td>0.162</td>
<td>0.002</td>
</tr>
<tr>
<td>CTI</td>
<td>51.5 ha</td>
<td>0.311</td>
<td>5.295</td>
<td>0.056</td>
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<tr>
<td>CUT (%)</td>
<td>51.6 km$^2$</td>
<td>0.143</td>
<td>0.103</td>
<td>0.007</td>
</tr>
<tr>
<td>CUT$^2$</td>
<td>51.6 km$^2$</td>
<td>-0.260</td>
<td>0.055</td>
<td>0.012</td>
</tr>
<tr>
<td>SHRB (%)</td>
<td>51.6 km$^2$</td>
<td>0.154</td>
<td>0.055</td>
<td>0.005</td>
</tr>
<tr>
<td>constant</td>
<td>NA</td>
<td>-6.859</td>
<td>1.925</td>
<td>&lt;0.001</td>
</tr>
</tbody>
</table>

Table 7. Predictive performance of individual resource models for predicting foraging activity of nine grizzly bears (odds ratio) compared to random available locations. Number ($n$) of field-visited telemetry observations with food resource use documented is reported.

<table>
<thead>
<tr>
<th>Food resource</th>
<th>$n$</th>
<th>Prob. of occurrence</th>
<th>Presence-absence</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Odds ratio</td>
<td>SE</td>
</tr>
<tr>
<td>sweet vetch (HEDY)</td>
<td>253</td>
<td>7.3</td>
<td>2.0</td>
</tr>
<tr>
<td>cow parsnip (HELA)</td>
<td>25</td>
<td>12.2</td>
<td>14.3</td>
</tr>
<tr>
<td>Hymenoptera (HYME)</td>
<td>204</td>
<td>7.0</td>
<td>1.8</td>
</tr>
<tr>
<td>buffaloberry (SHCA)</td>
<td>82</td>
<td>15.6</td>
<td>5.4</td>
</tr>
<tr>
<td>ungulate carcass/kill site (UNGL)</td>
<td>51</td>
<td>68.9</td>
<td>41.8</td>
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</table>
Table 8. Model coefficients describing the probability of female grizzly bear occupancy in southern Alberta.

<table>
<thead>
<tr>
<th>Variable†</th>
<th>Coef.</th>
<th>SE</th>
<th>p</th>
</tr>
</thead>
<tbody>
<tr>
<td>Agriculture</td>
<td>-7.982</td>
<td>1.718</td>
<td>&lt;0.001</td>
</tr>
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<td>Natural sub-region§:</td>
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</tr>
<tr>
<td>Alpine</td>
<td>5.208</td>
<td>0.722</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>Sub-alpine</td>
<td>4.399</td>
<td>0.402</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>Montane</td>
<td>1.685</td>
<td>0.266</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>Upper Foothills</td>
<td>1.536</td>
<td>0.191</td>
<td>&lt;0.001</td>
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<tr>
<td>Central Mixedwood</td>
<td>-1.679</td>
<td>0.386</td>
<td>&lt;0.001</td>
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<tr>
<td>Foothills Fescue</td>
<td>-0.226</td>
<td>0.673</td>
<td>0.737</td>
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<tr>
<td>Foothills Parkland</td>
<td>-0.099</td>
<td>0.672</td>
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<tr>
<td>Constant</td>
<td>-0.865</td>
<td>0.121</td>
<td>&lt;0.001</td>
</tr>
</tbody>
</table>

† All variables measured as proportions within a 10-km radius;
§ Lower foothills used as a reference (indicator) for natural sub-region categories; No female grizzly bears were observed in Dry Mixedwood, Central Parkland, Peace River Parkland, or Mixed Grassland natural sub-regions (occurrence therefore assumed at 0).

Table 9. Zero-inflated Poisson model results evaluating the relationship between realized habitat quality and occupancy-abundance of grizzly bears at hair-snag locations in west-central Alberta, Canada. Explanatory variables represent standard deviation changes in habitat quality.

<table>
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<th>Variable (scale)</th>
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<th></th>
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<tbody>
<tr>
<td></td>
<td>Coef.</td>
<td>SE</td>
<td>p</td>
</tr>
<tr>
<td>HQr (300-m)</td>
<td>0.813</td>
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</tr>
<tr>
<td>HQr (10-km)</td>
<td>0.318</td>
<td>0.115</td>
<td>0.006</td>
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<tr>
<td>constant</td>
<td>0.736</td>
<td>0.128</td>
<td>&lt;0.001</td>
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</tbody>
</table>

Group ~A (counts) | Coef. | SE   | p     |
<table>
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Figure 1. Study area depicting elevation, towns, Foothills (stippling) and Rocky Mountain (non-stippled area) natural regions.
Figure 2. Predicted distribution of major grizzly bear resource items by resource item group (a. presence of Hedysarum roots; b. herbaceous plant richness; c. animal matter; and d. fruit).
Figure 3. Representative seasonal (bi-monthly periods: a. 7-May; b. 7-July; c. 7-August) and multi-seasonal (d. 1 May to 31 September expressed as % of total possible) potential habitat quality.
Figure 4. Mean (±SE) seasonal (bi-monthly) selection expressed as odds ratios for a 1 unit standard deviation increase in potential habitat quality. Significance of bi-monthly periods based on a one-sample t-test (H₀: mean > 1) with level of significance indicated by star symbols. Number of bears tested in each season described below seasonal point estimates.
Figure 5. Patterns of multi-seasonal realized habitat quality (a) levels of habitat deficit (b).
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Figure 6. Average realized and potential multi-seasonal habitat quality (% of total) by landcover type (see Table 2 for codes). Habitat deficits (in absolute terms) reported above realized habitat landcover categories.
Appendix I. Importance weights (raw and scaled [in parentheses]) for 10 dominant food items represented in the diet of grizzly bears in west-central Alberta, Canada (adapted from Munro et al. 2006). Importance weights represent percent digestible matter and are presented by season (bi-monthly period) and ecosystem type (a. Foothills; b. Mountains).

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b. Mountains

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CHAPTER 9: GRIZZLY BEAR HEALTH – ENVIRONMENTAL RELATIONSHIPS

Marc Cattet\textsuperscript{11}, David Janz\textsuperscript{12} and Matt Vijyan\textsuperscript{13}

\textsuperscript{11}Canadian Cooperative Wildlife Health Centre; \textsuperscript{12}University of Saskatchewan, \textsuperscript{13}University of Waterloo

Introduction

The broad objective of this research component is to investigate relationships between landscape structure, human-caused landscape change, grizzly bear health and population performance through combined use of remote sensing technology, Global Positioning System (GPS) radio-telemetry, wildlife health evaluation, and molecular techniques in proteomics. Specific objectives of the project are to:

1. Enhance use of geospatial tools for monitoring landscape structure over large geographic areas, with particular emphasis on detecting changes likely to cause adverse health effects to resident grizzly bears, (this work is described in Chapter 4)

2. Determine home ranges and evaluate health of free-ranging grizzly bears captured throughout their distributional range in Alberta,

3. Develop and validate sensitive techniques, with emphasis on proteomics, for detecting long-term physiological stress in grizzly bears based on analysis of stress-activated substances found in blood serum, skin/muscle, and hair of grizzly bears,

4. Determine relationships between long-term physiological stress and other measures of health (longevity, growth, reproduction, immunity, and activity) in grizzly bears, and

5. Establish linkages between stress levels and health profiles of individual grizzly bears, performance of grizzly bear populations, and landscape structure and change within home ranges and population ranges along a gradient of human-caused alteration.

We have made significant progress on all objectives in 2008 and continue to meet the milestones outlined in the activity schedule of our original proposal. Specific details are:
Objective 1 – see Chapter 4.

Objective 2
The following summarizes activities completed related to the collection of grizzly bear home range and health data in 2008:

Study Area
The study area where all collaring efforts took place was discussed in Chapter 1.

Grizzly Bear Health Evaluation
We gathered health information from 20 of 23 grizzly bears captured in Alberta in 2008. Our most complete health information came from 14 bears captured specifically for this project, and include physical and physiological measurements recorded on site as well as results from laboratory analyses of blood serum, skin, and hair. We also collected skin biopsies (for stress protein array analysis) from 7 free-ranging bears by remote biopsy darting from helicopter, a technique less invasive than capture that holds promise for broad application in monitoring of population performance. All health data for 2008 are now entered into our project health database bringing the total number of health records for the project to 328 cases representing 201 unique animals of which 63 have multiple (2-8) records over intervals ranging from 1 month to 8 years.

Objective 3
Development and validation of long-term stress biomarkers is following two complementary paths. One path, led by Matt Vijayan (U. of Waterloo) is focussed toward blood serum-based indicators of long-term stress. The other path, led by David Janz (U. of Saskatchewan), is focussed toward development of a sensitive protein array and hair cortisol extraction procedure for detecting long-term physiological stress in skin/muscle and hair samples collected from grizzly bears.

At the U. of Waterloo, we are developing new tools to detect long-term stress/health status of wildlife. Specifically we are developing novel serum proteins as markers of stress/health status of grizzly bears. To this end, we have cloned and sequenced for the first time grizzly bear corticosteroid-binding globulin (CBG) and from the deduced amino acid sequence, generated a peptide sequence for antibody production. This affinity purified anti-bear CBG cross reacts very well with CBG in grizzly bear and polar bear sera. Using this antibody we have characterized CBG expression in serum of grizzly bears subjected to different capture stress and also in animals from different populations. We are in the process of developing an enzyme-linked immunosorbent assay (ELISA) for quantifying serum CBG concentrations in bear serum. In addition, we are also identifying differentially secreted proteins and/or peptide fragments in bear serum using proteomics. Using differential ingel electrophoresis (DIGE) we have identified several proteins that are differentially expressed in the serum of “stressed” bears. The majority of these proteins belong to the acute phase proteins, a family of proteins that are part of the innate immune response process, suggesting that they may be indicative of the health status of the animal. We are currently evaluating the utility of using heterologous antibodies (generated from human or mouse protein), that are commercially available, for
detecting the expression of these proteins in bear serum. We have identified transferrin as a key acute phase protein that is differentially expressed in serum of stressed bears. We are in the process of characterizing the utility of these acute phase proteins as markers of long-term stress/health status of grizzly bears. The work mentioned above is part of the PhD project of Mr. Brian Chow that involves identification and development of serum proteins as biomarkers of long-term stress in grizzly bears. We have completed measuring plasma cortisol, hsp70 and hsp60 levels in serum samples of grizzly bears for the past 8 years. Once the CBG ELISA and antibodies for acute phase proteins are characterized, we will quantify CBG and acute phase protein levels in these samples as well.

At the U. of Saskatchewan, we have utilized the antibody-based protein array that was initially developed in years 1-2 to determine expression of 31 stress-associated proteins in n=106 individual grizzly bear skin samples collected from throughout the species range in Alberta. In addition, we spent considerable time in year 3 further validating the protein array, including determination of stress proteins in skin from different body regions of bears, and examination of the effects of differing sample storage conditions (e.g., time to freezing of sample). Progress in this Objective is right on target as originally proposed in the Activity Schedule. Our focus is now on analyses of this large data set using the following general approaches: (1) further validation of the protein array by evaluating the biological relevance of changes in stress protein expression within and among bears (i.e., do the observed differences in stress protein expression within/among bears make biological sense?), (2) evaluation of relationships between stress protein expression and other measures of long-term stress and animal health (i.e., are there biologically relevant relationships among skin stress protein expression, serum stress protein expression, and integrated health indices such as growth?), and (3) evaluation of relationships between stress protein expression and human-affected landscape condition (i.e., are there differences in stress protein expression among bears occupying home ranges with differing human disturbance?).

Other than the data analyses described above, future work related to the protein array will follow two paths: (1) expanding the number of stress proteins included on the protein array, and (2) evaluate the utility of the array in other animal species to test its broad applicability in wildlife. In addition, we have developed a complementary technique to measure long-term stress in this project over the past year: determination of cortisol in hair samples collected from bears. The technique has been validated in the lab and bear samples are currently being analysed for cortisol levels to include in the statistical analyses in relation to other measures of long-term stress, health and landscape condition.

**Objectives 4 and 5**

Over years 1-3, the research team developed extensive datasets which describe the spatial configuration of landscape structure linked with detailed wildlife health information for grizzly bears across the distributional range of the species in Alberta, and the spatial configuration of annual landscape change (including anthropogenic-caused change) linked with wildlife health information for two provincial grizzly bear population units inhabiting this landscape during the same time periods. From our initial analyses of these
datasets performed during the first 6 months of year 3, we have strong evidence showing linkages between environment, long-term stress, and wildlife health. Although generally supportive of our working hypothesis (described in our proposal to NSERC), these relationships were often more complex than predicted due to inter-annual variations in bears’ resources and sex-age segregation across habitats. Further, because human activity was directly correlated with habitat quality, we were challenged in our analyses to separate the differential effects of these confounding factors.

We found the strongest environment-stress relationships were positive, suggesting that stress levels of grizzly bears were likely to increase as a result of human-caused changes to their home range over the preceding year as well as human activity at the time of capture. However, we also found evidence of weaker associations that were less conclusive, and call for future analyses with refinement of both spatial (home range) and temporal (period over which change is measured) scales.

We found evidence of relationships between stress levels of grizzly bears and their general state of health. For example, bears with high serum levels of heat shock protein 60 (i.e., high stress level) were in poorer body condition than bears with low serum levels (i.e., lower stress level), and this pattern was evident in all sex and age classes. In some cases, however, stress-health relationships could be explained alternatively as environment-health relationships. For example, a competing explanation to the relationship between heat shock protein 60 and body condition was that a higher density of roads in a bear’s home range results in lower body condition. These sometimes confounding effects between measures of environment and stress emphasize need for us to develop statistical analysis procedures that allow for modeling of more complex relationships among variables.

Since September 2008, we have been addressing difficulties encountered in our first round of analyses in preparation for a second round to be performed in March-April 2009. Specifically, we are modifying our datasets to allow more flexible merging of environmental and health data such that spatial and temporal scales can be adjusted in accordance with biological hypotheses. We are also improving our use of structural equation modeling to utilize repeated health and stress measures from individual bears, and to generally improve our modeling of complex relationships among environment, stress, and health variables. In addition to these improvements, we will be using substantially larger datasets in our next round of analyses that include for the first time stress protein array data (described under Objective 3), as well as spatial configuration of annual landscape change linked with wildlife health information for one more provincial grizzly bear population unit (bringing the total to three) inhabiting this landscape during the same time period.
APPENDIX 1: LIST OF PROGRAM PARTNERS (1999 – Present)

Ainsworth Lumber Co. Ltd.
Alberta Advanced Education and Technology (formerly Innovation and Science)
Alberta Conservation Association
Alberta Environment
Alberta Fish & Game Association
Alberta Newsprint Company
Alberta Sustainable Resource Development (formerly Natural Resources Service)
Alberta Tourism, Parks and Recreation
Anadarko Canada Corporation
Anderson Exploration Ltd.
AVID Canada
B P Canada Energy Company
BC Oil and Gas Commission
Buchanan Lumber – Tolko OSB
Canada Centre for Remote Sensing
Canadian Association of Petroleum Producers (CAPP)
Petroleum Technology Alliance Canada (PTAC)
Environmental Research Advisory Council (ERAC) Fund
Canadian Natural Resources Ltd.
Canfor Corporation
Center for Wildlife Conservation
ConocoPhillips Canada (formerly Burlington Resources Canada Ltd.) (formerly Canadian Hunter Exploration Ltd.)
Conservation Biology Institute
Daishowa Marubeni International Ltd.
Devon Canada Corp
Elk Valley Coal
Cardinal River Operations
Enbridge Inc.
Encana Corporation
Foothills Research Institute (formerly Foothills Model Forest)
Forest Resources Improvement Association of Alberta (FRIAA)
G&A Petroleum Services
GeoAnalytic Inc.
Government of Canada
Canadian Forest Service, Natural Resources Canada
Canadian Wildlife Service
Environment Canada – HSP
Human Resources and Skills Development Canada
Natural Sciences and Engineering Research Council of Canada (NSERC)
Parks Canada
Banff National Park
Jasper National Park
Grande Cache Coal Corporation
Hinton Fish and Game Association
Hinton Training Centre
Husky Energy Inc.
Komex International Ltd.
Lehigh Inland Cement Limited
Luscar Ltd.
Coal Valley Resources Inc
Gregg River Resources Ltd.
Manning Diversified Forest Products Ltd.
Manning Forestry Research Fund
Millar Western Forest Products Ltd.
Mountain Equipment Co-op
NatureServe Canada
Nexen Inc.
Northrock Resources Ltd.
Peregrine Helicopters
Persta
Petro Canada Ltd.
Peyto Energy Trust
Precision Drilling Corporation
Rocky Mountain Elk Foundation - Canada
Shell Canada Limited
Slave Lake Division – Alberta Plywood
Spray Lake Sawmills Ltd.
Suncor Energy Inc.
Sundance Forest Industries Ltd.
Talisman Energy Inc.
Telemetry Solutions
TransCanada Pipelines Ltd.
University of Alberta
University of Calgary
University of Lethbridge
University of Saskatchewan
Western College of Veterinary Medicine
University of Washington
University of Waterloo
Veritas DGC Inc.
West Fraser Mills Ltd.
Alberta Plywood
Blue Ridge Lumber Inc.
Hinton Wood Products
Slave Lake Pulp
Sundre Forest Products
Weyerhaeuser Company Limited
Wilfred Laurier University
World Wildlife Fund Canada
APPENDIX

APPENDIX 2: LIST OF PROGRAM PARTNERS (2008)

Alberta Tourism, Parks and Recreation
Alberta Sustainable Resource Development
B P Canada Energy Company
Canadian Association of Petroleum Producers (CAPP)
    Petroleum Technology Alliance Canada (PTAC)
    Environmental Research Advisory Council (ERAC) Fund
Canadian Natural Resources Ltd.
Canfor Corporation
ConocoPhillips Canada
Daishowa Marubeni International Ltd.
Devon Canada Corp
Elk Valley Coal
    Cardinal River Operations
Encana Corporation
Foothills Research Institute
Forest Resources Improvement Association of Alberta (FRIAA)
Grande Cache Coal Corporation
Government of Canada
    Human Resources and Skills Development Canada
    Parks Canada
    Jasper National Park
Husky Energy Inc.
Manning Forestry Research Fund
NatureServe Canada
Petro Canada Ltd.
Shell Canada Limited
Suncor Energy Inc.
Sundance Forest Industries Ltd.
Talisman Energy Inc.
TransCanada Pipelines Ltd.
West Fraser Mills Ltd.
    Blue Ridge Lumber Inc.
    Hinton Wood Products
    Slave Lake Pulp
    Sundre Forest Products
Weyerhaeuser Company Limited
APPENDIX 3: LIST OF PUBLISHED PAPERS


Schwab, B.L. 2003. Graph Theoretic Methods for Examining Landscape Connectivity and Spatial Movement Patterns: Application to the FMF Grizzly Bear Research. MSc Thesis. Department of Geography, University of Calgary, Calgary AB.


