

## Section 3: Climate and Plant Phenology

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### 3.1 Introduction

Climate change is altering habitats for a wide breadth of organisms with plants being particularly sensitive given their limited ability to physically follow suitable environmental conditions (Parmesan 2006). While this vulnerability to plants is notable in itself, it also has implications for species that are dependent on plant resources for their habitat. This leaves open the possibility for extinction cascades to be initiated via bottom-up effects, thus increasing vulnerability for species at higher trophic levels which rely on the availability and health of primary producers. Therefore, climate change vulnerability assessments for species at higher trophic levels, such as large mammals, may need to be based on assessments of plant communities and the food web interactions with mammals, under the assumption that food webs related to the target species are consistent in time and environmental space. While such trophically-based vulnerability assessments may be rare in the literature (but see Tuanmu et al. 2012), they are well suited to wide-ranging species with habitats largely defined by the existence of food plant communities including large mammals.

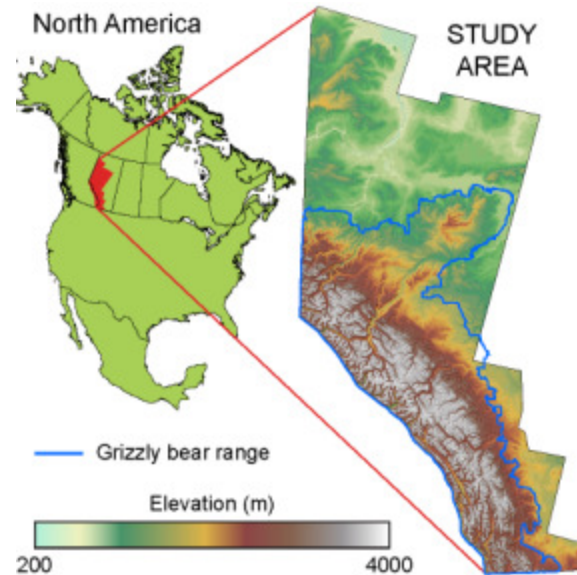
For the North American grizzly bear (*Ursus arctos*), the high energetic requirements of winter hibernation combined with the relatively short foraging season means that habitat tends to be selected based largely on resource availability (Schwartz et al. 2003). While faunal food sources such as ungulate predation are important energy sources, bears also dedicate much effort to foraging on vegetation, particularly in temperate forested habitats (Bojarska and Selva 2012) where isotopic diet assessments demonstrate that grizzly bears in the southern Canadian

Rockies are among the most vegetarian of North American populations (Hilderbrand et al. 1999, Mowat and Heard 2006). In spring and to a lesser extent throughout the active period, root digging for species such as *Hedysarum alpinum* (alpine sweet vetch), dominate the diet and thus the activity of bears (McLellan 1990, Hamer et al. 1991, Munro et al. 2006). As spring progresses with green-up, succulent herbaceous foods high in protein, such as young *Equisetum* sp. (horsetails) and *Trifolium* sp. (clovers) become more important. And finally, during the late-summer and early autumn hyperphagic period, bears in the southern Canadian Rocky Mountains rely predominantly on fruit from perennial shrubs for energy (Munro et al. 2006).

The threat of anthropogenic footprints and activities, such as roads, urban development, and resource extraction, which can all adversely affect grizzly bear survival rates, have been well-described (Mattson 1990, McLellan 1990, Nielsen et al. 2006, 2008). However, little is known about how climate change may affect the foods and thus the habitats that bears depend on. A variety of empirical models of grizzly bear habitat have been developed. These models have focused primarily on parameterizing current bear habitat (but see Nielsen et al. 2008), mostly using complex environmental surrogates of food resources and human activity such as land cover type, forest crown closure, slope, aspect, elevation, and metrics of road density or urbanisation (e.g. Nielsen et al. 2006, Ciarniello et al. 2007, Graves et al. 2011). Although some studies have focused directly on food resources on which bears depend (e.g. Nielsen et al. 2003, Nielsen et al. 2004, Nielsen et al. 2010), few have assessed how climate change would affect those resources and thus grizzly bear habitat (see, however, the recent review by Butler (2012) on potential linkages between climate and bear foods). First, woody encroachment of

subalpine and alpine meadows threatens sweet vetch habitat and other important alpine sources of bear foods. Second, decreased winter snowpack may limit the number of new avalanche slopes, which also represent quality foraging habitat for bears. Rodriguez et al. (2007) presents the only study to our knowledge of changes in grizzly bear food source consumption in relation to changing climate conditions over a thirty year period in Spain. While the authors found relationships between changes in autumn bear diets and changes in climate regimes over the same period, changes in land use and grazing practices were also noted.

In this study, we assess climate change vulnerability to grizzly bear food plant species using field data on bear food distribution and ecological niche models. Specifically, we project habitat changes for 17 species considered to be key grizzly bear food resources in the southern Canadian Rocky Mountains of Alberta and British Columbia (Figure 3.1). To assess risk levels to habitat stability, we also examined seasonal changes in species prevalence across the southern Canadian Rocky Mountains and identify geographic trends of habitat changes. While food species prevalence or probability of occurrence is not an absolute proxy for bear habitat quality, model predictions allow us to make inferences regarding general patterns among species and geographic space of future climate-related vulnerabilities to key grizzly bear food resources.



**Figure 3.1** Location of the study area in the western Canadian Rocky Mountains, showing topographic profile. Present-day grizzly bear range for the southern Canadian Rockies (Nielsen et al. 2009) is outlined in blue.

## 3.2 Materials and Methods

### 3.2.1 Ecological niche models

Ecological niche models relate environmental predictors with the response of observed species presence or absence, using any of a number of statistical procedures (Elith and Leathwick 2009). We incorporate eight individual modeling methods into a single averaged ensemble output, which has been shown to improve accuracy (Araújo and New 2007, Roberts and Hamann 2012a): artificial neural networks, classification tree analysis, generalized additive models, generalized boosting models, generalized linear models, multivariate adaptive regression splines, a boosted regression tree known as Random Forest, and surface range envelopes (not included in the ensemble) (Thuiller 2003). All modelling methods, as an output variable, report a probability of presence (PoP) for each species.

Species niches were parameterized in the models using species presence-absence data from 7,088 field vegetation plots located throughout the study area from three independent data sources. 4,090 vegetation plots were established in 1977-1979 within Banff National Park, Kootenay National Park, Jasper National Park, and Yoho National Park (see Nielsen et al. 2003 for details), 2,849 plots were sampled between 2001-2008 in west-central Alberta by Nielsen et al. (2010), and 149 plots were sampled in the Willmore Wilderness Area in 2001-03 (Gould 2007) and 2009-10 (Gould, unpublished).

Environmental predictors for each plot included nine seasonal and annual climate variables, three topographic/radiative variables, and one remotely sensed variable of forest crown closure. Climate data were generated for the 1961-1990 historic period using PRISM down-sampling (Daly et al. 1994) via a publically available software package that generates monthly, seasonal, and annual climate variables (Wang et al. 2012). Of the available climate variables, we selected ten which covaried least, including five temperature and precipitation metrics: mean annual temperature, average winter temperature, average summer temperature, winter precipitation, and summer precipitation; two dryness indices (Hogg 1997): annual heat moisture and summer heat moisture; and two calculated temperature-based metrics: the number of frost free days and the number of degree days above 5°C. Topographic/radiative predictor variables were included in models since these were previously found to be important in describing species distributions for bear foods in the region (Nielsen et al. 2003, Nielsen et al. 2010). Topographic predictors included a compound topographic index that incorporates moisture and drainage (Moore et al. 1991), a topographic heat load index that measures annual

solar radiation with a lagged heat load effect for southwestern aspects using slope, aspect, latitude (McCune 2007), and finally a general slope/aspect topographic radiation based only on slope and aspect using the topographic radiation aspect (TRASP) index of Roberts and Cooper (1989) within the Geomorphometric and Gradient Metrics ArcGIS Toolbox (Evans 2011). Last, we included a remotely sensed metric of forest crown closure from McLane et al. (2009), as forest structure has been identified as an important local predictor of occurrence for our species of interest (Nielsen et al. 2010).

### 3.2.2 Model Validation

Ecological niche models were validated using a random cross-validation of training points, where two thirds of training points were used to build models and the remaining one third used to evaluate model projections. While cross-validations have shown to be optimistic measures of model accuracy (Araújo et al. 2005), they have also been shown to be effective in selecting among modelling techniques (Roberts and Hamann 2012a). Cross-validation points were evaluated by calculating the area under the curve (AUC) of the receiver operating characteristic (Fawcett 2006). The AUC provides a threshold-independent evaluation of true presences vs. false presences for all probability of presence outputs simultaneously, where an AUC value of 1.0 is a perfect match and where random PoP data would produce an AUC of 0.5.

To explicitly quantify uncertainty in projections of the models (e.g. Buisson et al. 2010, Beale and Lennon 2012), we present a measure of model-based uncertainty based on the standard deviation of the multi-method projections incorporated into an ensemble projection. We calculated the standard deviation of the projected PoP from all individual modelling methods included in the ensemble (mean) calculation. The rationale for this calculation is that increased

divergence between individual model projections represents increased uncertainty in the ensemble projection. It has been demonstrated that the choice of statistical modelling method can represent a large component of overall variability in species projections (Dormann et al. 2008, Buisson et al. 2010, Mbogga et al. 2010) and we attempt to quantify that variability here. Similar metrics have been used by Gray & Hamann (2011), using rates of agreement in projections of binary presence and absence, and by Roura-Pascual et al. (2009).

### **3.2.3 Species habitat projections**

Present day and future species' habitats were projected at 300m x 300m resolution, with environmental predictor data generated as described above. Future climate grids were generated by calculating the arithmetic average of 14 general circulation model (GCM) outputs as temperature and precipitation anomalies from the present day. The effectiveness of averaged multi-GCM climate projections has been questioned (Fordham et al. 2011), as using multi-model averages of climate has the effect of centralizing more extreme climate projections. However, it also has the advantage of not unknowingly selecting an overly-extreme or overly-conservative climate scenario. To bookend the extent of potential future warming, two emissions scenarios, the aggressively warming A2 scenario and the moderately warming B1 scenario (IPCC 2007), were also considered.

Apart from the climate variables, future predictor data were left unchanged from present-day values, with the exception of estimates of crown closure. Projections of future crown closure were generated using a separate niche modelling procedure, incorporating the same predictor variables as described above for the species models, but using only the Random Forest bootstrapped regression tree method (Breiman 2001). Crown closure models were evaluated

by training the model with data from the northern half of the study area and validating models with data from the southern half, to simulate a projection into warming climates. The correlation between modeled and observed crown closure in the southern half of the study area was calculated to be  $r=0.49$ . Areas of agricultural use, surface water, and high alpine rock and ice were removed from the species habitat projections, as they were considered unsuitable future habitat. These areas are not considered in any of the summary analyses presented here. For the presence-absence maps, these locations have been masked as absences and do not affect model validation since training plots and thus validation points were not located in these areas.

To convert all other PoPs to binary presence-absence outputs, a threshold PoP was selected for each species. Threshold criteria that balance sensitivity and specificity have been shown to be more accurate than arbitrary thresholds or those based on maximizing Cohen's kappa statistic, especially when the prevalence rates of the species in question tend to be low (Jimenez-Valverde and Lobo 2007). Sensitivity-specificity balancing methods tend to favor, however, errors of commission in the data (i.e. favoring sensitivity, or potential overestimates of species presence). To select a threshold for presence-absence delineation from the PoP data, two methods were employed and the resulting values averaged: 1) the PoP which maximized the sum of sensitivity and specificity, and 2) the PoP which minimized the difference between the absolute values of sensitivity and specificity. Contrary to the findings of Jimenez-Valverde et al. (2007), threshold values for the presence-absence calculation methods were not correlated with the prevalence of species in the sample plot data, which was used for validation and randomly sampled for model training.



While maps show species projections for the entire study area, summary statistics of elevation profile and change in total distribution (area) were completed only for data within the boundary of present-day grizzly bear habitat (as defined by Nielsen et al. 2009) in order to quantify threats to bear foods for their currently inhabited range (shown in blue in Figure 3.1).

#### 3.2.4 Software

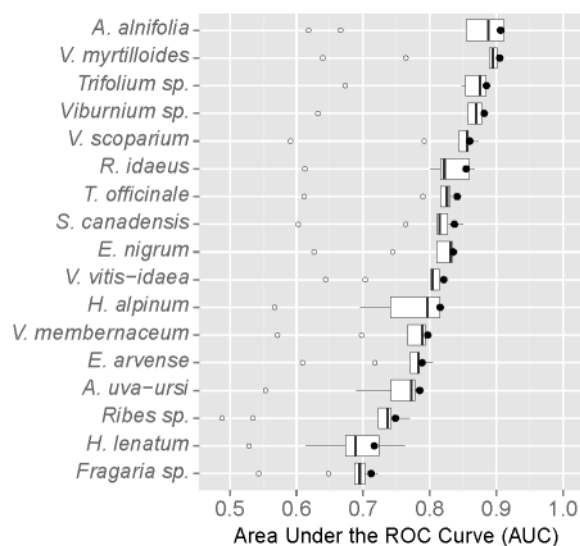
All modelling, analysis of results, and generation of output files were done in the R programming environment (R Core Team 2009). All eight individual models were generated with the *BIOMOD* package (Thuiller et al. 2009), AUC and associated statistics were calculated using the *ROCR* package (Sing et al. 2005), the PCA was performed using the base stats package, and rasters and graphics were generated using the *raster* package (Hijmans and van Etten 2012) and the *ggplot2* package (Wickham 2009).

### 3.3 Results

#### 3.3.1 Model Validation

AUC for all methods within each species is shown in Figure 3.2. All AUC values for all species and all methods, including the number of observed presences and absences, are provided Table 3.1. With the exception of the surface range envelopes, which showed very poor model performance (median AUC=0.61; maximum AUC=0.68), model validations indicated good to excellent model fit for all methods, with median AUC values ranging from 0.74 for the classification tree method to 0.83 for Random Forest. The ensemble method of averaging outputs from all other methods performed better than any individual method (median AUC=0.84). While it has previously been shown that including even poor-performing models in the ensemble calculations may increase the accuracy of the ensemble (Roberts and Hamann

2012a), we did not find this to be the case in our analysis. Including the surface range envelope outputs in the ensemble decreased its accuracy and was thus omitted from the calculation. AUC values for individual species for the ensemble method ranged from 0.72 for *Fragaria virginiana* (wild strawberry) to 0.91 for *Amalanchier alnifolia* (saskatoon).

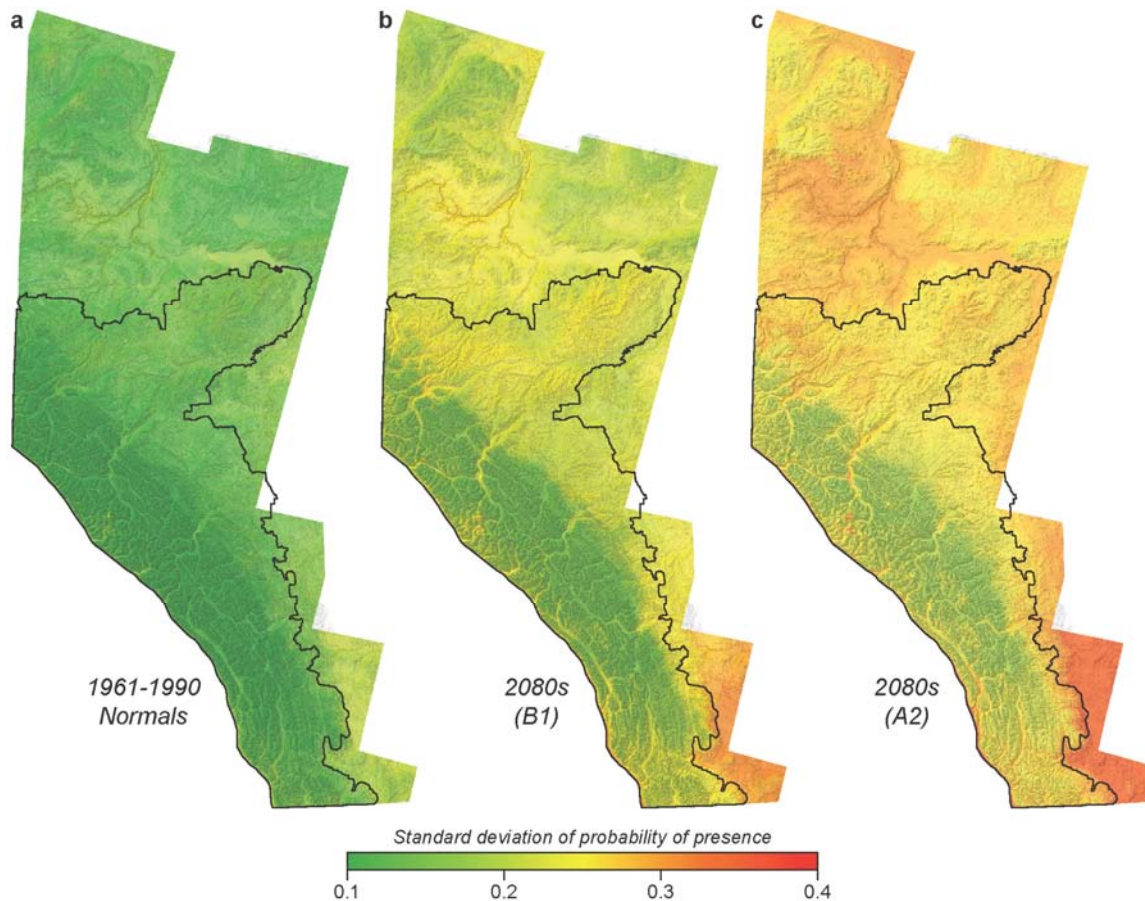


**Figure 3.2** Boxplot showing model accuracy as measured by the area under the curve of the receiver operating characteristic (AUC) for each species and all modelling methods, including the ensemble mean method (MEAN), the AUCs for which are shown as black dots. A complete table of AUC values for each species within each method and number of observed presences and absences is provided in Table 3.2.

Within present-day grizzly habitat, PoP standard deviations tended to be low to moderate, with the greatest model agreement occurring in the higher mountainous areas, suggesting that future climate conditions here may be well-represented in the training data. The highest model disagreements in both B1 and A2 scenarios appear in the low prairies in the southern extent of the study area, along the bottoms of some southern river valleys, and in some of the warmer and drier low elevation areas in the north (see maps in Figure 3.3). This suggests that projected climates without modern analogues in the training data may be influencing model divergence in these locations.

**Table 3.1** Area under the curve of the receiver operating characteristic (AUC) for all modelling methods, including the ensemble mean method which omits the SRE method (MEAN). The number of presence (Npres) and absence (Nabs) records in the training data sample p.

| Species                        | N <sub>pres</sub> | N <sub>abs</sub> | ANN  | CTA  | GAM  | GBM  | GLM  | MARS | RF   | SRE  | MEAN | Species median |
|--------------------------------|-------------------|------------------|------|------|------|------|------|------|------|------|------|----------------|
| <i>Amelanchier alnifolia</i>   | 82                | 2261             | 0.87 | 0.67 | 0.91 | 0.89 | 0.91 | 0.85 | 0.89 | 0.62 | 0.91 | 0.89           |
| <i>Arctostaphylos uva-ursi</i> | 446               | 1897             | 0.74 | 0.69 | 0.77 | 0.74 | 0.78 | 0.78 | 0.78 | 0.56 | 0.79 | 0.77           |
| <i>Empetrum nigrum</i>         | 193               | 2150             | 0.81 | 0.75 | 0.83 | 0.81 | 0.84 | 0.83 | 0.83 | 0.63 | 0.84 | 0.83           |
| <i>Equisetum arvense</i>       | 419               | 1924             | 0.77 | 0.72 | 0.78 | 0.78 | 0.78 | 0.78 | 0.80 | 0.61 | 0.80 | 0.78           |
| <i>Fragaria virginiana</i>     | 743               | 1600             | 0.69 | 0.65 | 0.70 | 0.69 | 0.70 | 0.69 | 0.72 | 0.55 | 0.72 | 0.69           |
| <i>Hedysarum alpinum</i>       | 233               | 2110             | 0.70 | 0.74 | 0.79 | 0.80 | 0.81 | 0.82 | 0.82 | 0.57 | 0.82 | 0.80           |
| <i>Heracleum lanatum</i>       | 79                | 2264             | 0.69 | 0.61 | 0.70 | 0.73 | 0.67 | 0.68 | 0.76 | 0.53 | 0.72 | 0.69           |
| <i>Ribes</i> sp.               | 253               | 2090             | 0.49 | 0.72 | 0.74 | 0.74 | 0.74 | 0.74 | 0.77 | 0.54 | 0.75 | 0.74           |
| <i>Rubus idaeus</i>            | 158               | 2185             | 0.80 | 0.82 | 0.83 | 0.86 | 0.82 | 0.82 | 0.87 | 0.62 | 0.86 | 0.82           |
| <i>Shepherdia canadensis</i>   | 650               | 1693             | 0.83 | 0.77 | 0.81 | 0.81 | 0.82 | 0.81 | 0.85 | 0.61 | 0.84 | 0.81           |
| <i>Taraxacum officinale</i>    | 195               | 2148             | 0.83 | 0.79 | 0.83 | 0.83 | 0.82 | 0.82 | 0.83 | 0.62 | 0.85 | 0.83           |
| <i>Trifolium</i> sp.           | 155               | 2188             | 0.87 | 0.85 | 0.88 | 0.89 | 0.88 | 0.85 | 0.88 | 0.68 | 0.89 | 0.88           |
| <i>Vaccinium membranaceum</i>  | 335               | 2008             | 0.77 | 0.70 | 0.79 | 0.77 | 0.79 | 0.79 | 0.80 | 0.58 | 0.80 | 0.79           |
| <i>Vaccinium myrtilloides</i>  | 108               | 2235             | 0.89 | 0.77 | 0.90 | 0.90 | 0.89 | 0.89 | 0.90 | 0.64 | 0.91 | 0.89           |
| <i>Vaccinium scoparium</i>     | 426               | 1917             | 0.86 | 0.80 | 0.86 | 0.84 | 0.86 | 0.84 | 0.87 | 0.60 | 0.87 | 0.86           |
| <i>Vaccinium vitis-idaea</i>   | 512               | 1831             | 0.80 | 0.71 | 0.81 | 0.80 | 0.81 | 0.80 | 0.83 | 0.65 | 0.83 | 0.80           |
| <i>Viburnum</i> sp.            | 203               | 2140             | 0.86 | 0.82 | 0.88 | 0.87 | 0.87 | 0.86 | 0.88 | 0.64 | 0.89 | 0.87           |
| <i>Method median</i>           |                   |                  | 0.80 | 0.74 | 0.81 | 0.81 | 0.82 | 0.82 | 0.83 | 0.61 | 0.84 |                |



**Figure 3.3** Standard deviations of the ensemble mean species projections of probability of presence, averaged across all species, as a measure of model uncertainty. The black outline represents modern grizzly bear habitat in the southern Canadian Rockies (Nielsen et al. 2009).

### 3.3.2 Changes in species distributions

Complete summaries of habitat gained and lost for each species are shown in Table 3.2. Maps of PoP and plots of the distribution for each species along the observed elevation gradients for the modern and future periods are presented in Appendix C. Within present-day grizzly bear range, most species (with some notable exceptions) experienced increased PoPs in the 2080s projections for both the B1 and A2 warming scenarios relative to the 1961-90 period. PoPs also typically increased from the B1 to the A2 scenario suggesting a trend of increased habitats with increased warming. When PoPs were converted to presence or absence, most species

demonstrated a corresponding increase in projected range for the 2080s period for both scenarios (again, typically larger in the A2 than B1 scenario). Saskatoon gained the most suitable habitat, increasing in projected area by 186% and 199% in the B1 and A2 scenarios, respectively. *Vaccinium membranaceum* (Huckleberry) and *Vaccinium myrtilloides* (Blueberry) also showed substantial increases in habitat by 84-112% and 66-78%, respectively. Notable exceptions to increased trends include *Vaccinium scoparium* (grouseberry), which was projected to lose 92-95% of its habitat, *Empetrum nigrum* (crowberry), which was projected to lose 72-89% of its habitat, and wild strawberry, which was projected to lose 30-58% of its habitat. The remaining species gained or lost projected habitat in more moderate amounts ranging from losses of 18% to gains of 70%.

In addition to the area of total habitat by species, amount of stable habitat was also summarized within present-day grizzly bear range. The measure of stable area represents the proportion of a species' 1961-90 habitat that remains suitable habitat in future projections (Table 3.2). While most species were projected to maintain between 94-100% of their 1961-90 habitat, some species were projected to lose substantial amounts of habitat. *E. nigrum*, *V. scoparium*, *Fragaria virginiana*, and *Arctostaphylos uva-ursi* (bearberry) in both scenarios, and *H. alpinum* in the A2 scenario all lost considerable amounts of stable habitat (retaining between 3% and 67%). *H. alpinum* in the B1 scenario shows an increase in habitat area of 45% but a loss of stable area (only 56% stable), indicating that, while net habitat area may increase, just over half of the present-day range of the species remains suitable habitat in future projections.

**Table 3.2 Summary of species' projected elevation and area changes within occupied grizzly bear habitat (Nielsen et al. 2009) for the 2080s period for two emissions scenarios: B1 (moderate warming) and A2 (aggressive warming). Median elevations (Elev) for the 1961-1990 period and the projected changes ( $\Delta$  Elev) in 10th percentile (p10), median (p50), and 90th percentile (p90) elevation under each future climate scenario are listed, as are total range area (Area) and percent change in range area ( $\Delta$  Area) for the 1961-1990 observed climate and the two 2080s climate projections. Percent stable area (Stable area) represents the proportion of the species' 1961-1990 range that is maintained in the projections for the 2080s. Trends of species' seasonal use by bears (Season) are based on findings of Munro et al. (2006).**

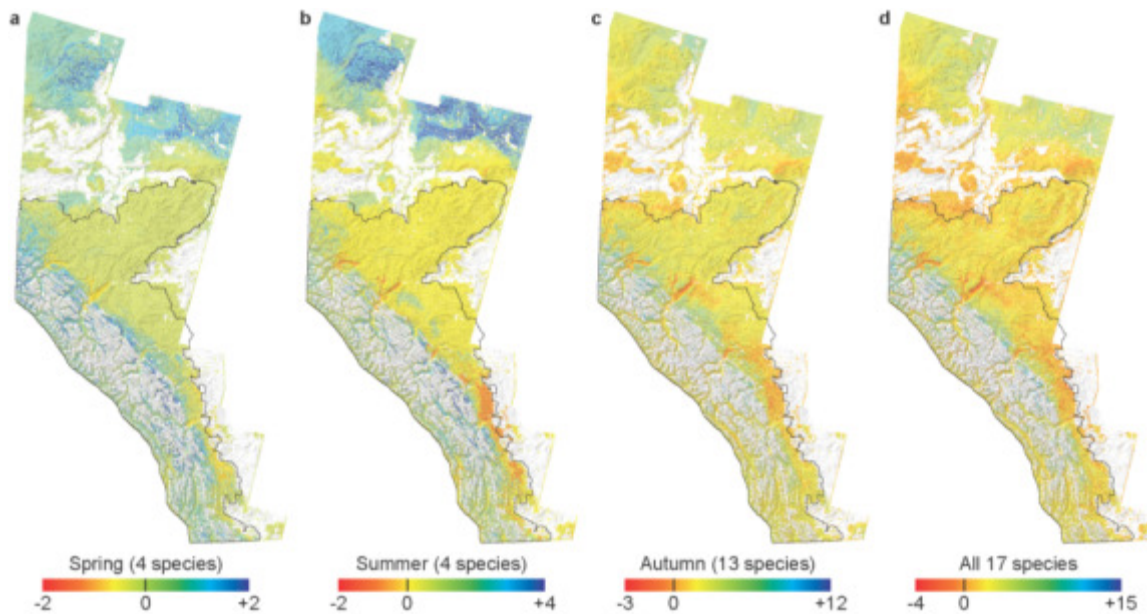
| Species                        | Common       | Season   | Elev (m) |      |      | $\Delta$ Elev. (m) |      |        | Area       |      |        | $\Delta$ Area                     |       | Stable area |       |      |
|--------------------------------|--------------|----------|----------|------|------|--------------------|------|--------|------------|------|--------|-----------------------------------|-------|-------------|-------|------|
|                                |              |          | 61-90    |      |      | 2080s (B1)         |      |        | 2080s (A2) |      |        | 1961-90                           | 2080s |             | 2080s |      |
|                                |              |          | p10      | p50  | p90  | p10                | p50  | p90    | p10        | p50  | p90    | ( $\times 10^5$ km <sup>2</sup> ) | B1    | A2          | B1    | A2   |
| <i>Amelanchier alnifolia</i>   | saskatoon    | Aut      | 880      | 1378 | 1822 | (-57)              | -96  | (+91)  | (-51)      | -68  | (+153) | 4.47                              | +186% | +199%       | 100%  | 100% |
| <i>Arctostaphylos uva-ursi</i> | bearberry    | Aut      | 1096     | 1497 | 1995 | (-136)             | -57  | (+70)  | (-109)     | -101 | (+49)  | 5.42                              | -11%  | -18%        | 67%   | 58%  |
| <i>Empetrum nigrum</i>         | crowberry    | Aut      | 1587     | 1901 | 2176 | (+242)             | +193 | (+181) | (-993)     | -767 | (+201) | 2.89                              | -72%  | -89%        | 22%   | 3%   |
| <i>Equisetum arvense</i>       | horsetail    | Spr      | 766      | 1036 | 1360 | (+25)              | +96  | (+315) | (+34)      | +152 | (+513) | 6.94                              | +30%  | +45%        | 100%  | 100% |
| <i>Fragaria virginiana</i>     | strawberry   | Aut      | 794      | 1219 | 1776 | (+244)             | +212 | (+187) | (+440)     | +423 | (+329) | 9.71                              | -30%  | -58%        | 61%   | 34%  |
| <i>Hedysarum alpinum</i>       | sweet vetch  | Spr- Aut | 1219     | 1518 | 2111 | (+133)             | +357 | (+108) | (+449)     | +525 | (+207) | 1.37                              | +45%  | -15%        | 56%   | 31%  |
| <i>Heracleum lanatum</i>       | cow parsnip  | Sum      | 809      | 1237 | 1846 | (+20)              | +74  | (+131) | (+20)      | +74  | (+131) | 11.42                             | +17%  | +17%        | 100%  | 100% |
| <i>Ribes</i> sp.               | gooseberry   | Aut      | 808      | 1218 | 1733 | (+20)              | +92  | (+233) | (+21)      | +94  | (+245) | 11.34                             | +17%  | +18%        | 100%  | 100% |
| <i>Rubus idaeus</i>            | raspberry    | Aut      | 778      | 1080 | 1493 | (+23)              | +89  | (+169) | (+44)      | +192 | (+404) | 8.10                              | +28%  | +55%        | 99%   | 100% |
| <i>Shepherdia canadensis</i>   | buffaloberry | Aut      | 803      | 1459 | 1921 | (+33)              | -66  | (+87)  | (+19)      | -124 | (+61)  | 7.40                              | +42%  | +70%        | 97%   | 100% |
| <i>Taraxacum officinale</i>    | dandelion    | Spr-Sum  | 791      | 1127 | 1582 | (+15)              | +69  | (+221) | (+31)      | +153 | (+374) | 9.23                              | +16%  | +36%        | 99%   | 100% |
| <i>Trifolium</i> sp.           | Clover       | Spr-Sum  | 781      | 1099 | 1552 | (+11)              | +42  | (+107) | (+20)      | +80  | (+244) | 8.27                              | +13%  | +22%        | 99%   | 99%  |
| <i>Vaccinium membranaceum</i>  | huckleberry  | Aut      | 1139     | 1616 | 2035 | (-114)             | -111 | (+24)  | (-335)     | -203 | (-4)   | 4.96                              | +84%  | +112%       | 100%  | 100% |
| <i>Vaccinium myrtilloides</i>  | blueberry    | Aut      | 773      | 1046 | 1386 | (+49)              | +224 | (+480) | (+56)      | +265 | (+591) | 7.50                              | +66%  | +78%        | 100%  | 100% |
| <i>Vaccinium scoparium</i>     | grouseberry  | Aut      | 1634     | 1940 | 2239 | (+382)             | +288 | (+296) | (-414)     | +171 | (+319) | 2.11                              | -92%  | -95%        | 7%    | 3%   |
| <i>Vaccinium vitis-idaea</i>   | lingonberry  | Aut      | 792      | 1134 | 1711 | (+6)               | +24  | (+57)  | (+17)      | +85  | (+107) | 8.75                              | +13%  | +30%        | 94%   | 97%  |
| <i>Viburnum</i> sp.            | cranberry    | Aut      | 786      | 1105 | 1501 | (+37)              | +176 | (+390) | (+43)      | +205 | (+474) | 8.85                              | +44%  | +51%        | 100%  | 100% |

In addition to changes in projected area of habitat, all species showed changes in their elevation profile with a general future trend towards increasing median elevation, increasing 10<sup>th</sup> percentile elevation (the trailing downslope edge), and increasing 90<sup>th</sup> percentile elevation (the leading upslope edge) (Table 3.2). The largest median elevation increases were projected in the A2 scenario for *H. alpinum* and *Fragaria virginiana*, with increases of 525m and 423m, respectively. With only one exception, the 90<sup>th</sup> percentile elevation of all species increased in both emissions scenarios. Median and 10<sup>th</sup> percentile elevation decreases were projected for some species including *V. membranaceum* (median decrease of 203m in the A2 scenario), *A. uva-ursi* (median decrease of 101m in the A2 scenario), and *A. alnifolia* (median decrease of 68m in the A2 scenario). All three of these species are wide-ranging in their present-day elevation profile. *E. nigrum* was the only species exhibiting contradictory trends in elevation profiles for different emissions scenarios. Projected 10<sup>th</sup> percentile and median elevations for the species increased moderately in the B1 scenario (by 242m and 193m, respectively), but decreased substantially in the A2 scenario (by 993m and 767m, respectively).

### 3.3.3 Changes in distribution of seasonal resources

When summarized by foraging season for grizzly bears (as per Munro et al. 2006), the diet richness (number of species present) during spring (hypophagia), summer (early hyperphagia), and autumn (hyperphagia) periods tended to be either stable or increasing (Figure 3.4, Figure 3.5). It should be noted that these summaries reflect only the net balance of diet items (species counts) and do not reflect the loss of one species that is replaced by the arrival of another species (i.e. community turnover in diet composition). Springtime conditions showed no major change in diet richness for most of the present-day grizzly bear range with the exception of

some high mountainous areas that gain habitat for one species and for some regions along the eastern slopes and river valleys of the Rocky Mountains that lose a single species reflecting the loss of *H. alpinum* habitat (Figure 3.4). Grizzly bear habitat in the summertime is projected to maintain present-day diet richness or increase diet richness in the higher-elevation areas of the Rocky Mountains with up to four additional species. The only decrease in diet richness in summer occurs in the southernmost valley bottoms losing up to two of the four summer species considered. The same trends as in the spring and summer were also present during the fall: a general stability of diet richness within present-day grizzly bear range. Decreases in habitat of up to three species occurred in the eastern slopes, while increases in high elevation habitat of up to eleven of the twelve species considered.



**Figure 3.4** Maps showing the change from the 1961-1990 period in number of species projected present for the A2 scenario of the 2080s period. Changes are shown by season for (a) spring, (b) summer, and (c) autumn, with the seasonally-important bear food source species (see Table 1). (d) Changes in counts of all 17 species considered (all seasons together), is also shown. Areas of unsuitable range (agriculture, rock, ice, water, etc.) are masked in white. The range of present-day grizzly bear habitat in the southern Canadian Rockies is shown as a black outline. Maps of absolute counts of species are shown in Figure 3.5.



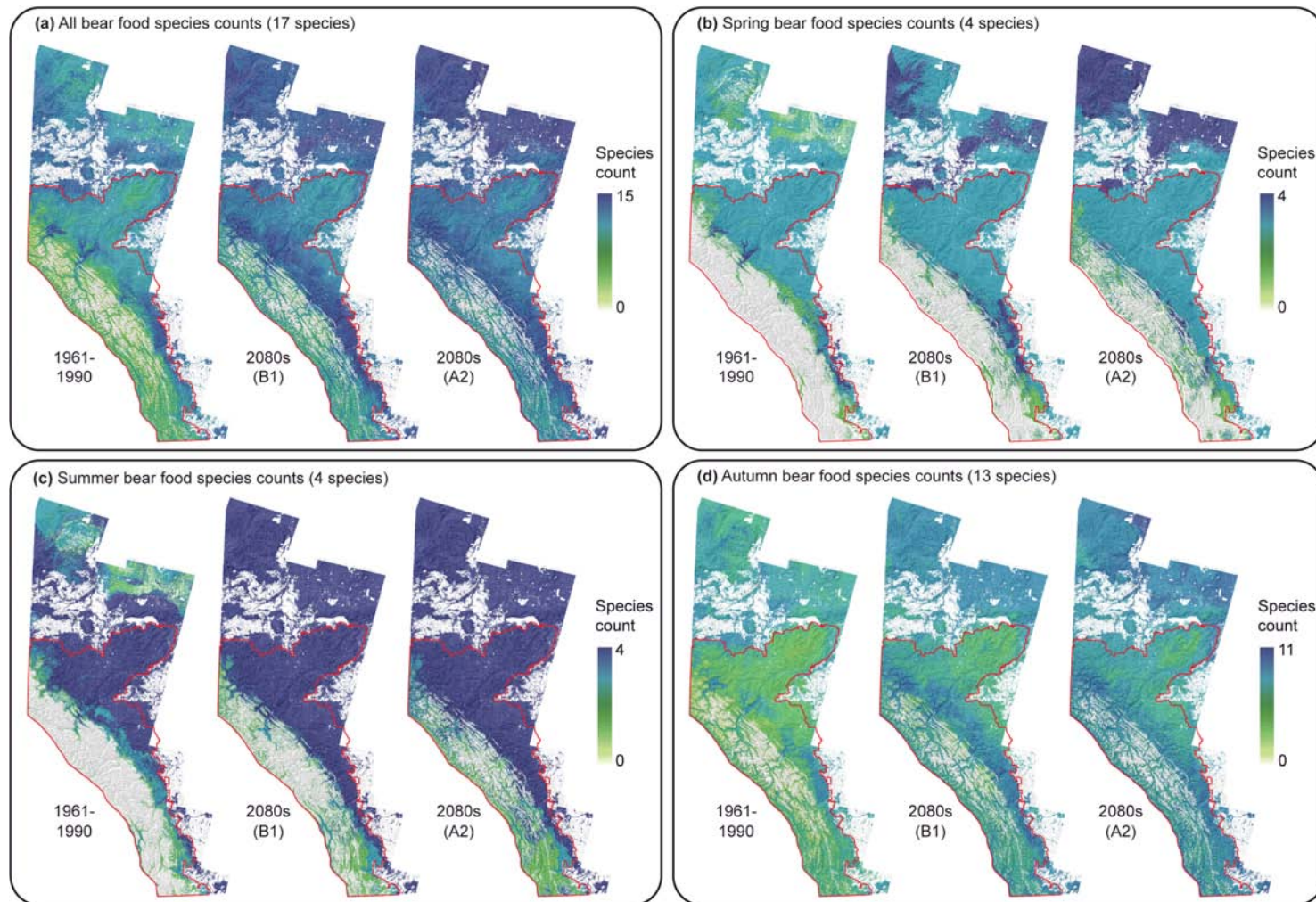


Figure 3.5 Maps showing counts of projected species for the 1961-1990 period and for the B1 and A2 scenarios for the 2080s period for (a) all species, as well as species identified as seasonally important for bears in (b) the spring, (c) the summer, and (d) the autumn. Areas of unsuitable range (agriculture, rock, ice, water, etc.) are masked in white. Present-day grizzly bear habitat in the southern Canadian Rockies is shown as a red outline.

## 3.4 Discussion

### 3.4.1 Climate change vulnerability of grizzly bear food items

The lack of an extensive loss from climate change in habitat for most plant-based grizzly bear food items suggests that widespread collapse of habitats due to losses in trophic web linkages on which bears depend is unlikely. In fact, the general trend within present-day grizzly bear range of the southern Canadian Rocky Mountains is towards more widespread availability and increased diet richness of bear foods. Changes in model projections which have high uncertainty could, however, alter responses. For instance, the general projection of increased diet richness ignores key species interactions such as competition, which may, despite environmental suitability for a new site, prevent future establishment (successful colonization).

Also related, our models assume an 'equilibrium and perfect dispersal' scenario where species are expected to reach all suitable habitats by the 2080s. Owing to steep elevation clines in the area, this may be a reasonable assumption for many species since emerging habitat would be geographically proximate to current habitat. However, some bear food items, particularly the ericaceous shrubs that are an important source of fruit (energy) to bears, are known to have low dispersal-establishment capacity with their long-term persistence in an area predominantly due to vegetative reproduction (Vander Kloet & Hill 1994; Regan et al. 2012). These species may only persist in habitats that today are projected to be within the environmental niche (i.e. zero dispersal scenario) of the species in the 2080s. Less topographically diverse locations may be vulnerable to the majority of species since the rate of migration would need to be higher, especially considering that these areas are more likely to be fragmented by anthropogenic barriers.

In addition to changes in specific locations of habitats, there were notable changes in the elevation profile of food items that have meaningful implications for bears. A general trend of uphill migration of species may increase habitat quality at higher elevations thus offsetting losses elsewhere and further providing a source population for rescue effects of sink populations. Given the propensity for human interactions to negatively affect bears and bear survival rates (Mattson 1990), the persistence of grizzly populations under such a scenario relies on the continued preservation of upslope habitats free from anthropogenic disturbance and human activity, as is a feature of the National Parks and other protected areas within current grizzly bear range. Additionally, many downslope locations such as lower foothills and valley bottoms are projected to lose species richness by as much as 50% for summer species or 25% of the seasonally-important hyperphagic season in late summer and autumn. This may increase human-bear conflicts since food may be more limiting in these areas.

While not entirely positive, our prognosis for the overall vulnerability of grizzly bears to climate change is markedly better than those for similar high trophic level mammals, including bears. Recent assessments of food resources for a population of giant pandas (*Ailuropoda melanoleuca*) in China project substantial reductions in availability of many bamboo species (Tuanmu et al. 2012), representing an acute threat to this species. As a North American example, the polar bear (*Ursus maritimus*) is a species of conservation concern that has enjoyed extensive public exposure in terms of its sensitivity to climate change effects (citation). Primary threats to polar bears from climate warming relate to habitat loss and losses of sea ice in particular, which can adversely affect hunting and breeding. Less sea ice coverage and earlier spring sea ice breakup results in less ideal seal hunting conditions, shorter on-ice feeding

periods, and more energy expended in swimming activities, leading to a general decrease in reproduction rates and litter sizes (Derocher et al. 2004). While Canadian grizzly bears do not face such challenges, the implications for all upper trophic level species is one of general uncertainty regarding the availability of food resources and habitat quality. A reiteration of the conclusions by Tuanmu et al. (2012) would serve well here: it is important to consider interspecific interactions when assessing effects of climate change.

#### 3.4.2 Species of Concern

Species considered in our study that could be labeled “of concern” take two forms: 1) non-critical bear food species that show major reductions and/or changes in habitat, and 2) critical bear food species that show moderate or major reductions and/or changes in habitat. In the first case, the species in question may itself be at high risk, but the effect of its loss on the overall quality of bear habitat may be minimal due to the availability of other food resources and low use by grizzly bears. In the second case, critical bear foods, such as those which form a substantial proportion of diets throughout the year or those which are highly important to bears in a given season, are projected to have at least marginally reduced range sizes or to have limited stable habitat between the present and future.

An example in the first case, *V. scoparium* shows the largest habitat loss of any species considered in the B1 and A2 scenarios for the 2080s (92% and 95% loss, respectively) and the smallest amount of stable habitat (only 7% and 3% for B1 and A2, respectively). While not a critical food source by itself, bears rely heavily on all forms of ericaceous shrubs during the autumn hyperphagic period. Other ericaceous species such as *V. myrtilloides*, *V. vitis-idaea*

(lingonberry), and the more critical *V. membranaceum* all show increases in habitat and nearly complete maintenance of their present-day ranges, thus placing them at “low risk”. Projections for *E. nigrum* are very similar to *V. scoparium*, with habitat losses up to 89% and amount of stable habitat as low 3%. Not surprisingly, these two species have the highest-elevation ranges in the present day of all species considered. This, in combination with the projected elevation changes for future periods, suggests that climate-related habitat for these species may be pushed out of alpine areas and “off” mountain tops entirely by future warming. Other high-elevation species considered, including *V. membranaceum*, do not exhibit the same increases in elevation and habitat losses under future climate scenarios. *V. membranaceum* is less elevation-restricted than either *V. scoparium* or *E. nigrum*, as is evident by their much lower 10<sup>th</sup> percentile elevations. This wider climate niche may make them less sensitive to changes, particularly in temperature.

*H. alpinum*, the other notable high-elevation species considered in our study, is of particular concern as *H. alpinum* root digging represents an important spring and autumn food source for bears (Munro et al. 2006). Like *V. membranaceum*, this species exhibits a less-restrictive elevation profile. However, like *V. scoparium* and *E. nigrum*, projected *H. alpinum* habitat within present-day grizzly bear range shows large increases in elevation in both future scenarios B1 and A2. However, the corresponding changes in area of habitat are not consistent between future scenarios, with *H. alpinum* gaining 45% area of habitat in B1, while losing 15% area in A2. This difference is explained by the increased amount of habitat lost along the east slopes of the Rocky Mountains in the A2 scenario, while new upslope habitat emerges in roughly equal amounts in both scenarios. Density plots of elevation for future climate scenarios (Appendix C),

show equal probabilities of high-elevation habitat in both scenarios but disappearing habitats of mid-elevation in the A2 scenario. The net result for *H. alpinum* is that, while available habitat may not be drastically restricted in the future, proportions of stable area suggest that half to two-thirds of populations will be required to physically migrate into new locations to be maintained with the landscape. While these migrations may only be short upslope distances, the simple necessity of physical migration for the survival of the majority of this species raises concern, especially given the species' critical importance as a bear food source.

Two less critical food sources for bears, *Fragaria virginiana* and *A. uva-ursi*, are predicted to decrease in range. In both cases, the loss of habitat under future warming is unexpected, as these species are widespread within the present-day study area as well as throughout western North America, suggesting a tolerance for warmer conditions such as those projected for our future study area. In these cases, the projected habitat losses could be a result of the limits of the model training data, which may not adequately capture the full climate niche of these species, particularly the warmer end of the niche that would be represented to the south of our study area. This is particularly visible in wild *Fragaria virginiana*, which loses extensive habitat through the warmest areas of its present-day range through the centre of the province. By contrast, future projections for *A. uva-ursi* suggest substantial loss of habitat in the low-elevation river valleys throughout the Rocky Mountains with no emergence of higher-elevation habitats, as is the case with other widespread species such as *Taraxacum officinale* (dandelion). This would be an expected trend if the range of *A. uva-ursi* is not temperature driven, but rather defined by other environmental predictors, which may be the case.

### 3.5 Conclusion

This first look at changes to grizzly bear plant food resources under anthropogenic climate warming provides reasons for optimism, but also reasons for concern. With only a few exceptions, most species considered in this study appear to be at low risk of extirpation or even of widespread habitat instability. In fact, many species show the potential to increase in range under future climate warming. This would suggest that the majority of plant-related grizzly bear food items should continue to be available to bears within their current range through the coming century. We have, however, found a few species of concern with several species showing severe range losses under future climate warming. High-elevation, restricted species such as *E. nigrum* and *V. scoparium*, both of which are projected to lose nearly all their range within present-day grizzly bear habitat, should be considered at high risk. The effect of the loss of these individual species on bear habitat quality is, however, difficult to assess, given the maintained availability of most other autumn fruit resources and the fact that these fruit are less commonly used by bears. Of greater concern to bear habitat quality are the high rates of projected habitat loss and a lack of stable habitat for *H. alpinum*, a spring and autumn food resource for grizzly bears when little else is available. This presents a particular concern for grizzly bear conservation, given the overall importance of this species in their diets.

It should be noted that this study does not consider food quality (i.e. energetics) or local abundance, which may also be affected in unpredictable ways by anthropogenic alterations in the climate regime. We also did not consider how climate change may affect sources of animal protein that bears rely on including ungulates and ants, although generally warmer climates for cool mountain environments would be expected to benefit these species, particularly ants.

Future investigations into how climate change will affect quality and abundance of these food resources is needed to fully understand the impacts of climate change on grizzly bear habitats.

### **3.6 Acknowledgements**

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