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Ya Ha Tinda Elk Herd and Red Deer River Valley Ecotone Study: Final Report

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DISCLAIMER

This progress report contains preliminary data from ongoing academic research directed by the University of Alberta that will form portions of graduate student theses and scientific publications. Results and opinions presented herein are therefore considered preliminary and to be interpreted with caution, and are subject to revision.

COVER PHOTO CREDITS

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EXECUTIVE SUMMARY

This report serves to summarize: 1) data collected as part of a long-term monitoring of the grassland vegetation, population demography, and distribution of the Ya Ha Tinda (YHT) elk herd, 2) an analysis of long-term grassland change and current grassland ecotones, and 3) an update on the effects of summer aversive conditioning on elk distribution. The overall goal of these efforts are to provide information for understanding and managing trophic relationships in the Red Deer River Valley (RDRV) system in and adjacent to Banff National Park under natural disturbance regimes as part of Parks Canada management of the area. We report results since 2005 and discuss their management implications, as well as on-going and future efforts. Results prior to 2005 are reported in Hebblewhite 2006.

Long-term trends in woody plant encroachment onto grasslands in the Red Deer River drainage around YHT-West Lakes and Scotch Camp-Tyrrell Creek were quantified from aerial photography. With the exception of YHT, there was a 47 ± 12% decrease in grassland extent across sites between 1952 and 1992 with a corresponding increase of 39 ± 10% in conifer forest. The region of greatest grassland loss was at the West Lakes site where since 1952 there has been a 74% grassland loss corresponding to infilling of coniferous forests. At YHT there was a 9% decrease in grassland extent between 1952 and 1982 followed by a net increase of 9% from 1982 to 2003, which we attributed to shrub mowing.

Plant biomass and composition along grassland ecotones were sampled in July 2009 along 87 100-m transects of the grassland-shrub-forest ecotone to characterize the ecotonal changes associated with woody encroachment of grassland at four sites (Ya Ha Tinda (YHT), West Lakes (WL), Tyrrell Flats (TY) and Scotch Camp (SC)) in the Red Deer River Valley. Transects characterized vegetation within 5 community types and along different transition zones between plant community types. Within vegetation types, graminoid biomass was greatest in grassland and mixed-wood forest with graminoid biomass being lower in grasslands at WL than at YHT; forb biomass was higher at YHT than at WL. Shrub biomass was similar across sites being lowest in grasslands and highest in the shrublands, and most diverse in mixed-wood forests. Shrublands in the lower RDRV were dominated by bog birch (*Betula glandulosa*), whereas in the upper RDRV, willows species (*Salix* spp.) were dominant at most sites. Graminoid composition changed more than biomass along ecotones, while forb diversity declined from open to closed canopied vegetation and both shrub composition and biomass shifted along ecotones. Shrub overstory had a greater effect on understory biomass in shrublands than did shrub and tree overstory in forest communities.

Long-term grassland production was monitored at 45 sites from 2005-2010 to characterize annual and seasonal trends in plant growth and forage availability to elk. Peak biomass occurred in August. Summer precipitation was not strongly related to August current annual growth (CAG) likely because summer precipitation during 2005-2010 was less variable compared to much lower precipitation in 2002 and 2003. These data will be used in winter 2012 to derive MODIS-based estimates of productivity to assess the importance of YHT grassland production to the annual elk population growth rate from 2001-2011.

In terms of elk distribution and habitat use at the Ya Ha Tinda, we found that elk use of fuel-reduction treatments and burns increased over time since the treatment, but that patterns differed strongly between seasons. In the summer, elk showed high use of burned grasslands and burned cutblocks. In winter, elk avoided burned grasslands in 2009 and showed highest use of grasslands in winter 2010, but used burned forests by winter 2011, perhaps because of lower snow conditions. There was little use of burned cutblocks immediately after the fire (winter 2010) but by winter 2011, they were the third most used land cover type. Comparatively, deer also showed the highest use of burned grasslands during summer, but avoided burned forests and burned cutblocks. During the winter, deer were rare at Ya Ha Tinda. Whether or not white-tailed deer, especially, are increasing on the YHT can only be addressed using long-term pellet data.

The results of recent aversive conditioning (AC) were similar to Spaedtke (2009). We showed a strong shift away from the YHT grasslands, that echoed Spaedtke (2009)'s conclusions that AC may have had some success at increasing long-distance movements, if not necessarily migration per se. Our home range analysis approach did not allow us to test for differences in elk distribution after the cessation of AC in 2011. However, contrary to Spaedtke (2009), who reported a strong seasonal shift of elk away from grasslands during his study (2005-2007), we found weaker evidence across the entire AC time period (2005-2010) for a strong shift in spatial distribution away from targeted grasslands at the center of the YHT pastures. This discrepancy may be because of several reasons including methodological factors, but ultimately, ongoing analysis of the response of elk distribution to the cessation of aversive conditioning in the next 5 years will provide the strongest insights into whether aversive conditioning was successful at reducing elk use of grasslands at Ya Ha Tinda.

Overall, demographic evidence from aerial surveys, calf recruitment, adult female survival and pregnancy rates during the past 10 years suggest that the Ya Ha Tinda elk herd reached a low-density equilibrium of ~500 elk by 2005 and as of 2010 appeared to be held there by high predation rates by a diversity of predators. However, by the time of the completion of this report, there was some additional evidence the herd was continuing to decline. Wolves continued to be the leading cause of mortality, followed closely by human harvest by legal hunting (during the early 2000's) and First Nations harvest. Migrant to resident ratios appear to have stabilized around 1:1 between 2005-2010 as well, indicating that about 50% of the population is migratory in 2011. The mechanisms for this appear to be density-dependent increases in resident, but not migrant, adult female survival rates, and perhaps some density-dependent increase in calf recruitment that because of the higher migrant pregnancy rates could be driven by increasing migrant calf survival.

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1.0 Background

Since 2000, a collaborative program has been on-going between researchers at the Universities of Alberta and Montana, Parks Canada, Alberta Conservation Association and other natural resource groups within Alberta to determine how changes in the Ya Ha Tinda (YHT) elk population are affected by humans (harvest, habitat management), natural factors (predation, climate), and natural vegetation dynamics. In addition, Alberta Fish and Wildlife has collected long-term population data since 1972 on elk population size and calf recruitment. Our focus has been on understanding the changing migratory behavior of elk and the trophic dynamics within this predator-prey-montane grassland system. More recently, we have added a long-term perspective on grassland dynamics. Combined, our studies of the elk population at Ya Ha Tinda represent one of the longest elk population studies in a system with intact natural predators, including wolves (*Canis lupus*) and grizzly bears (*Ursus arctos*), and human hunting.

This report focuses on data collected during a 3-year from 2009-2011, a period funded by Parks Canada and Alberta Conservation Association, but also includes longer term data analyses and updates. During the specific 3-year funding period, the major objectives were: 1) a short-term study of the long-term vegetation change in the Red Deer River Valley (RDRV), and 2) continuation of the long-term monitoring of elk demography, distribution, and vegetation in support of management of the Ya Ha Tinda elk herd.

2.0 Study Area

The study area is the Red Deer River drainage within the Canadian Rocky Mountains of Alberta and includes areas within Banff National Park (BNP) and adjacent provincial lands. The region is classified as montane along valley bottoms and transitions to lower subalpine in areas above 1650 m (Holland and Coen 1983). Annual precipitation for Banff National Park is 475 mm, averaged from 1900 - 2007. Total mean precipitation in summer (1 May to 31 Aug) is 285 mm and 190 mm in winter (1 Nov to 30 Apr) (Fig. 2.1, Environment Canada, 1900 - 2007). Mean daily summer temperature is 12^{o} C and mean daily winter temperature is -5^{o} C. Combined with low precipitation, Chinook winds keep the grasslands at YHT mostly snow-free in winter (Stelfox 1981).

Soils in the Red Deer River Valley range include Brunisol and Luvisol under lodgepole (*Pinus contorta*) pine stands, with Humic Regosols, Melanic Brunisols (soils with >10 cm of Ah horizon) or Chernozemic soil in open grassland regions within both BNP and provincial lands (McGillis 1977, Holland and Cohen 1983). The grasslands of the Red Deer River Valley are described as *Festuca-Stipa richardsonii* associations (Looman 1969). Common grasses include foothills rough fescue (*Festuca campestris*), Junegrass (*Koeleria macrantha*), Hooker's oatgrass (*Helictotrichon hookeri*), and invasive smooth brome (*Bromus inermis*) (Willoughby 2001). Forested regions consist mainly of lodgepole pine, trembling aspen (*Populous tremuloides*),

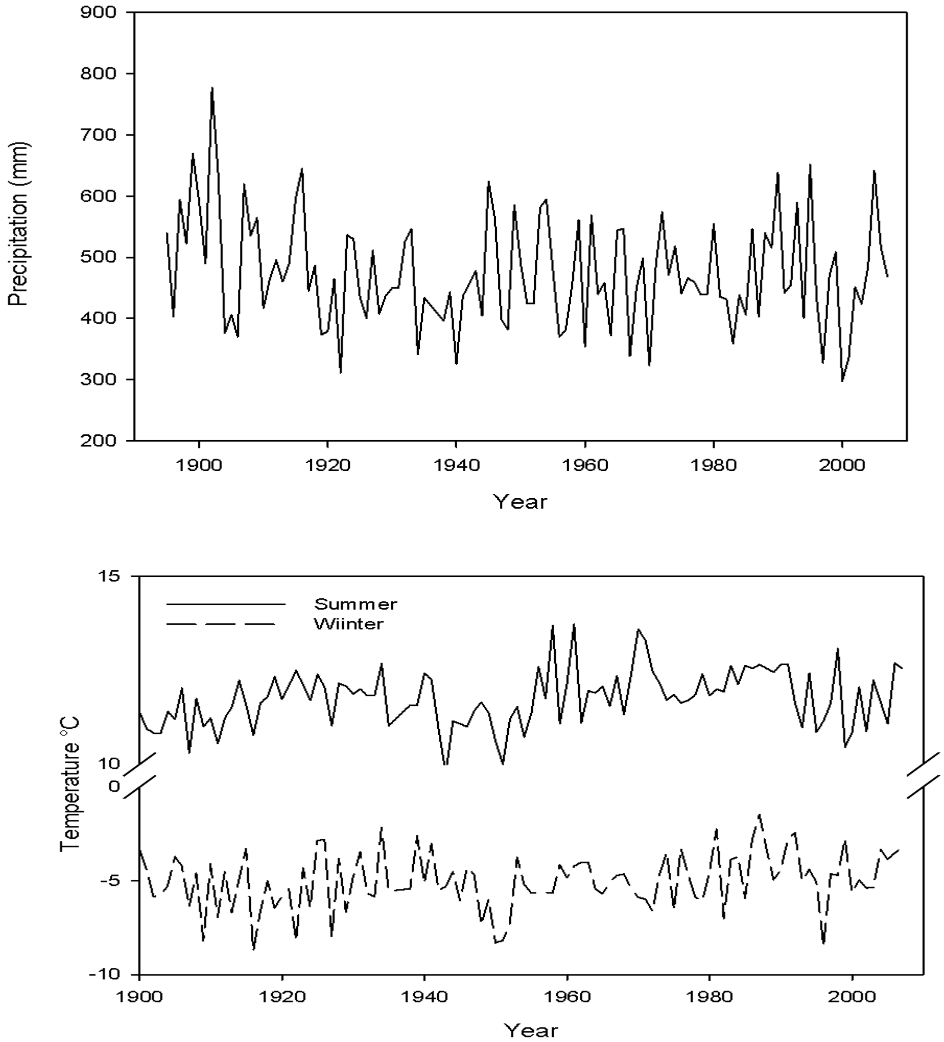


Fig. 2.1 (a) Annual total precipitation and (b) mean daily summer (May – Aug) and winter (Nov - Apr) temperature (° C) recorded at Banff CS weather station 1900 to 2007, Alberta, Canada.

balsam poplar (*Populus balsamifera*) and white spruce (*Picea glauca*). Common shrubs include bog birch (*Betula glandulosa*), shrubby cinquefoil (*Potentilla fruticosa)* and willow (*Salix* spp.*)* (McGillis 1977).

Ungulates using the Red Deer River Valley in the recent past include elk (*Cervus elaphus nelsoni*), deer (*Odocoileus virginiana* and *O. hemonius)*, bighorn sheep (*Ovis canadensis*) and moose (*Alces alces*). The area is not primary habitat for mountain goats (*Oreamnus americanus*) or woodland caribou (*Rangifer tarandus caribou*) (Stelfox 1993). Nineteenth century journals suggest that bighorn sheep and plains bison (*Bison bison bison*) were historically the most numerous species in the area, with elk and deer being encountered less frequently (Kay et al. 1994). Plains bison were extirpated by the 1870s (Soper 1941) before the government of Alberta put in place effective hunting restrictions to protect them (Stelfox 1993). Historically, caribou in BNP numbered thirty to forty in the 1980s, but the herd dwindled to ~5 members, and then effectively none, following the death of 3 females in an avalanche in spring of 2009.

Rocky Mountain elk populations (*Cervus canadensis*) have increased within BNP and the Red Deer River Valley since the early 1900s. Early in the twentieth century, elk populations were reduced to remnant herds in the Rocky Mountains and foothills (Millar 1916, Dwyer 1969). During a period of elk reintroductions from 1917 - 1920, 251 elk from Yellowstone National Park were translocated into BNP. The population in BNP grew and elk culls were initiated in 1941. Elk numbers in BNP prior to the 1940s are uncertain (Morgantini 1988). By 1969, when the herd reduction program ceased, it was postulated that a major migratory route had been established along the Red Deer River leading from BNP to the YHT (Morgantini 1988). Outside BNP, the first recorded observation of elk in the Panther-Red Deer-Clearwater region was reported on the Ya Ha Tinda ranch in 1933 (Cowan 1943). Cowan (1943) estimated the BNP population to be 4,000 wintering elk. McGillis (1977) reported 50 - 60 bulls on and adjacent to YHT in 1942 - 43, with the nearest band of cows wintering at the head of the James River to the northeast. By 1950, Flook (1970) reported that approximately 100 cows wintered on the YHT, and by 1961 1,400 – 1,800 individuals were observed in late winter (Webb 1961). The winter population at the YHT peaked in 1992 and has since declined (Fig. 2.2). The decline has been attributed to a number of causes including translocations (1994 - 1999) and the recovery of the wolf population (Hebblewhite 2006).

The elk herd wintering in the Red Deer River drainage was reported to migrate 25-50 km into BNP to summer (Gunson 1997), but no real population estimates of the number of migratory animals existed until 1973 when aerial surveys began (Fig. 2.2; Morgantini 1988, Skjonsberg 1993, Hebblewhite et al. 2006; A. Hubbs, Alberta Fish and Wildlife, unpublished data). The majority of elk summering in BNP were found in mountain regions, widely dispersed over some 1,600 km^2 of terrain (Morgantini 1988). Elk were found mostly in high subalpine and alpine ranges at the headwaters of several creek tributaries to the Red Deer, Panther and Clearwater rivers. In the 1970s and 1980s there was intermittent and low to moderate winter use of low elevation meadows, such as Scotch Camp (SC) and Tyrrell Creek (TY) in the upper Red Deer River drainage with elk typically using these sites as staging areas en-route between summer and winter ranges (Morgantini 1988, Hebblewhite et al. 2006). Elk use of SC and TY meadows in BNP was markedly higher during the late season hunt in 1977 and 1978 (Morgantini and Hudson 1985). However, shortly after the hunting seasons, the same animals re-established their traditional range at YHT. Summer use of the YHT grassland complex has increased since the early 1990s with the ratio of resident to migrant elk increasing times four since the late 1970s, despite a decline in the overall population (Hebblewhite et al. 2006). In response, Parks Canada initiated an aversive conditioning program in 2005 that resulted in the shift of range use to areas east of the YHT, evident particularly during 2005 - 2006 (Spaedtke 2009) and the period of this report (see section 7).

Fig 2.2. Ya Ha Tinda winter elk population (1973 - 2010) within the Red Deer River Valley, Alberta, Canada.

Predators within the study area include wolf (*Canis lupus),* grizzly bear (*Ursus arctos*), and cougar (*Felis concolor*) *(*Morgantini 1988). Wolves were considered abundant prior to European settlement at the turn of the twentieth century. Poisoning and shooting of wolves for livestock protection and hunting bounties led to extirpation in the southern regions of Alberta, as well as in BNP by the 1950s (Gunson 1992, Musiani and Paquet 2004). Re-colonization of wolves in BNP first occurred in 1985. Cougars have also been persecuted historically but are now increasing in the region (Stelfox 1993, Knopff 2010). Another major predator of elk is the grizzly bear, whose predation is primarily limited to springtime.

3.0 Long-term Trends in Woody Plant Encroachment onto Grasslands in the Red Deer River Valley

3.1 Background

Over much of the past century, fire suppression in the eastern slopes of the Rocky Mountains has resulted in an unnatural mean fire return interval and fire cycle (Rogeau 2002). Intentional burning of montane grasslands had long been carried out by aboriginal peoples to initiate spring growth and promote grassland productivity (Christenson 1971, Arno 1980, Kay et al. 1994). Canopy cover of woody species has increased in areas normally dominated by herbaceous and grass species where burning has not periodically occurred, changing the foothills landscape from pre-European settlement (Rhemtulla et al. 2002, Burkenshaw 2005). With canopy closure and shrub invasion, forage production can be reduced by as much as 60% (Johnson and Smoliak 1968) and plant composition altered to shrub-dominated vegetation including bog birch and willow (Willoughby 2001). The objective of Parks Canada is to maintain ecological integrity of the Park (Parks Canada 2007). The alteration of landscape level processes and shifts in plant species abundance can pose a threat to ecological integrity unless there is additional management (Parks Canada 2007).

An ecologically important component of the Red Deer River Valley is the montane fescue (*Festuca campestris,* Rydberg) grasslands. Fescue grasslands are highly productive ecosystems, which are now considered threatened in Alberta (Kemper 2009). It is estimated that 90% of fescue grassland has been greatly or moderately modified (Looman 1969), with more recent estimates indicating only 5% may be remaining in historical condition (Vujnovic 1998). Fescue grasslands provide important winter forage for herbivores due to its winter-hardy structure under snow and overall palatability (Willms et al. 1986). Loss of rough fescue grasslands have been attributed to grazing and woody plant encroachment (McGillis 1987). Montane rough fescue can be affected negatively by spring and summer grazing if there is little time for regrowth, resulting in shifts in dominance to other herbaceous species (Willms et al. 1985, Willms 1991, Merrill 2007). The increasing grazing pressure by a growing portion of the YHT elk herd remaining as residents in summer was one of the major motivations for establishing exclosures and conducting detailed grazing studies (McInenly 2004, Merrill et al. 2007) as well as for the aversive conditioning project (Spaedtke 2009, see section 7). The establishment of shrubs and trees can occur with a decrease in fire frequency resulting from removal of grass litter for fuel under heavy grazing (Van Auken 2003). A reduction in area or quality of grassland habitat due to encroachment of shrubs and trees could alter habitat capacity for a number of grassland species, including ungulates.

The objective of this research was to use aerial photographs to quantify the change in spatial extent, rate of change, and shift in patch sizes of grasslands during the period of 1952-2003 within the Red Deer River Valley. Aerial photographs were not available earlier than the 1950s. The four sites under study provide seasonal forage for elk in winter and along seasonal migration routes (Morgantini 1988).

3.2 Methods

Study Sites

The change in grassland characteristics was compared across four study sites along the Red Deer River (Fig. 3.1, Table 3.1, Appendix I). Sites were located along an east to west gradient decreasing in herbivory, shortened growing season (Hebblewhite 2006), and increasing gradient in moisture.

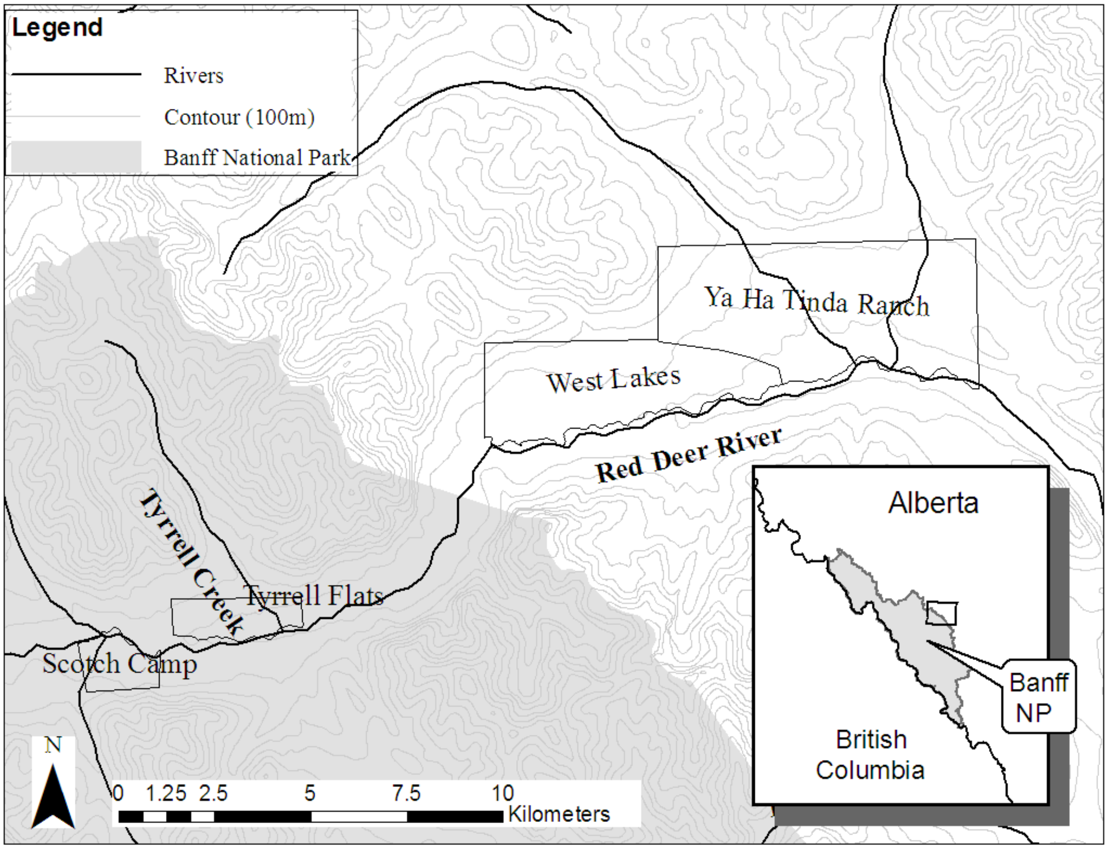


Fig. 3.1. Grassland study sites along the Red Deer River, Alberta, Canada. 2 sites, Scotch Camp (SC) and Tyrrell Creek (TY), were located inside the boundary of Banff National Park, and 2 sites, West Lakes (WL) and Ya Ha Tinda (YHT), were adjacent. See Appendix I for site boundaries.

Table 3.1. Physiographic and climatic characteristics of 4 grassland study sites, Scotch Camp (SC), Tyrrell Creek (TY), West Lakes (WL), and Ya Ha Tinda (YHT), along the Red Deer River, Alberta, Canada, including their grazing and burn history, and years of aerial photos used in analysis.

	SC	TY	WL	YHT	Source
Physiographic					
Area (ha)	249	329	1455	2634	7
Elevation (m)	1700-1800	1600-1700	1600- 1800	1600-1800	10
Soils	Brunisols Regosols	Brunisols Regosols	Regosols Brunisols	Brunisols Regosols	1, 2
Climatic					
Total annual precipitation (mm) 1984-2007	375	375	610	610	8
Grazing History					
Pre-1890	Low	Low	Low	Low	3, 4
1890 to1940	Low	Low	Low	Low	3, 5
1940 to 2003	Moderate	Moderate	High	Very high*	5, 6
Burning History (Appendix II)	Prescribed burns	Prescribed burns	Suppression	Suppression	9
Shrub control	--	--	--	1974, 1982, 2000s°	
Photo dates	1952,1962, 1972,1982, 1992	1952, 1962, 1972, 1982, 1992	1952,1962, 1972,1982, 1992, 2003	1952,1962, 1972,1982, 1992, 2003	10

* > 10 elk/km^2 considered high for YHT (White 2001, White et al. 1995)

[1] McGillis (1977), [2] Holland and Coen (1983), [3] Gunson (1977), [4] Kay et al. (1994), [5] Morgantini (1988), [6] White et al. (1995), [7] Glines (2011). Hebblewhite (2005), [8] Environment Canada Stations at SC and WL, [9] Parks Canada, Government of Alberta, [10] Parks Canada, NAPL, Alberta SRD.

° Incomplete records for 2000s.

The Ya Ha Tinda (YHT), which is the largest and most eastern site of the 4 study sites (Fig. 3.1, Table 3.1), is located 5 km east of the BNP boundary. West Lakes (WL) is contained within the YHT ranch boundaries, but was considered a separate study site due to lower elk densities over time than the YHT site (Morgantini 1988, Hebblewhite et al. 2006). By the late 1950s, Flook (1957) identified heavy grazing by the elk population on Ya Ha Tinda and predicted continued heavy use would result in rapid range deterioration. In 1970, there was an attempt to control shrubby cinquefoil at the YHT using Picloram, which was one of the first attempts at shrub control on the ranch (Scotter 1974). In 1976, attempts were also made to control bog birch encroachment by Picloram and Dowco (McGillis 1977, Trottier and Milner 1978). Mowing of bog birch was initiated in 1982 (Parks Canada 1987). Attempts to reseed native grasses occurred during the 1980s in select elk high-use areas such as Scalp Creek, and in a formerly non-native grass pasture seeded to *Bromus* spp., but attempts failed (Walker 1984). By the late 1980s, Seel and Wiebe (1989) concluded the net effect of high elk and horse grazing at the YHT was altering grassland composition and productivity. Fescue communities where grazing was excluded recovered as early as the first two years (McInenly 2003).

The WL site was not burned since 1929 and the YHT site since 1936 until recently (Appendix II). In 2007, there was a prescribed fire that burned 139 ha north of the YHT ranch buildings and up Hat Mountain in spring (Apr to June), and a much larger burn starting in March 2009 that burned a large portion (1774 ha) of the WL and YHT sites until mid-July 2009 (Table 3.1, Appendix II). The Scotch Camp (SC) and Tyrrell Creek (TY) meadows historically were spring and fall migration staging areas for elk migrating between winter range at YHT and BNP (Morgantini 1988, Skjonsberg 1993), and they have been described as low use areas for elk in winter (Morgantini 1988; Skjonsberg 1993). Prescribed burns have occurred periodically at SC and TY since 1983 (White 2001). There is also an east-west topographic and climatic gradient, which appears to influence grassland phenology (Hebblewhite et al. 2008).

Land Cover Classification Scheme

We classified each study site using six vegetated land cover types (coniferous forest, mixed-wood forest, shrubland, grassland, clear-cut, and pasture) and three un-vegetated land cover types (pond/river, road, and rock/bare ground). Vegetated land cover classes corresponded with the dominant canopy cover. Grasslands lacked woody plant cover (<30%), except for shrubby cinquefoil, which was considered a grassland community co-dominant (Willoughby 2001). Shrublands were defined when woody cover comprised >70% of the site. Woody shrub cover consisted largely of bog birch at YHT and WL and contained higher proportions (maximum of ~50% of shrub cover) of willow (*Salix* spp.) at the SC and TY sites. The mixed-wood forest cover type included areas with >60% forest canopy and deciduous trees (trembling aspen, balsam poplar) comprising >20% of total tree canopy cover. Coniferous stands (>60% forest) were composed of >80% lodgepole pine and/or white spruce. One area (21.7 ha) at the YHT site was clear-cut in 1987 then seeded to creeping red fescue *(Festuca rubra*) post-harvest, was classified as clear-cut in 1992 and 2003. In addition, a pasture area (13 ha) that contained natural grasslands seeded with tame brome grass (*Bromus* spp.) was established at YHT prior to 1952 (Morgantini 1995, Willoughby 2001 Sumners and Archibold 2007) and classified as pasture thereafter.

Aerial Photo Imagery and Pre-processing

Vertical aerial photographs of the study area were obtained from the Alberta Sustainable Resource Development (ASRD) air photo library in Edmonton, AB, for the years 1952, 1972, 1982, 1992, and 2003 (Table 3.1). The National Air Photo Library of Natural Resources Canada (NRCan) in Edmonton, AB, provided images for 1962 and 1972. Imagery for 2003 was available only for YHT and WL, and so analyses for this date were restricted to these sites. The 1952, 1992, and 2003 images were taken on 31 July, 10 August and 30 September, respectively, and were at a scale of 1:40,000. The 1962 imagery was taken on 15 June at a scale of 1:25000; the 1972 photographs were taken on 5 October at a scale of 1:21120 for the YHT and WL, and 27 August at a scale of 1:80000 for TY and SC. The 1982 imagery for all sites was taken on 2 September at a scale of 1:60000.

Because the imagery across years was of variable scale (1:25000 to 1:80000), it was scanned at a range of 600 to 2000 dots per inch (DPI) to obtain the same pixel resolution (1 m^2) among years based on the conversion of: scale/DPI. The 1952, 1962, 1972, 1982, 1992, and 2003 photos were co-registered to 2003 orthorectified satellite images in ArcInfo (ESRI, Redlands, CA) using 6-10 ground control points (GCP) per photo, which was consistent with Rhemtulla et al. (2002) for similar studies in Jasper National Park (JNP). GCPs were distributed evenly across the photo surface to ensure good fit throughout the study area (Morgan et al. 2010). Ground control points included roads and trail intersections and buildings. Spline and polynomial transformations were used for photo sets depending on which transformation produced the best fit to the orthorectified 2003 image base on root mean square error (RMSE). All images regardless of transformation technique had a RMSE <2 m, which is consistent with other high-resolution photo-interpretation studies (Bowman et al. 2001, Bai et al. 2005, Fairfax et al. 2003).

Photo Interpretation of Land Cover Types

The smallest unit of measure for delineating cover types, i.e., minimum mapping unit (MMU), was set at 500 m^2 at a 1:4,000 view scale for all photographs. Classification of land cover types within the study area was completed using a supervised classification of the land cover types based on visual interpretation (Morgan et al. 2010). The photo interpretation descriptions of cover types are included in Appendix III. Because the focus of this study was on native vegetation change, anthropogenic features (e.g., roads, buildings), rocky bare surfaces, and water bodies were lumped into a category called “Other”. These types were relatively stable over time, with minor (~0.2%) increases in the coverage by roads.

Classification Accuracy

To provide an assessment of the classification, the classification of the 2003 aerial photograph was compared to a classification based on 94 ground-truth locations visited in 2009 consisting of the two endpoints of a 100-m transect (Appendix IV). This assessment was conducted only at the WL and YHT sites because these were the only sites with 2003 imagery. While it would have been preferable to construct the error matrix using photographs taken closer to the sampling period of the ground-truth plots (i.e., 2009), these did not exist. Two types of accuracy assessment were derived. Producer accuracy (PA) is the probability that a pixel is classified correctly, while the user's accuracy (UA) is the probability that a pixel classified as a type on a map matches what is on the ground (Pontius and Lippitt 2006).

Change in Land Cover Extent and Patch Size

The total area of land cover types at each study site was calculated by summing the area of each polygon within a land cover type. The total area in land cover types was expressed as both extent (ha) and percent change in extent over time as $[((ha_{t+1} - ha_t)/ha_t) \times 100]$ where t is the extent in 1952 and ha_{t+1} is the extent in the year of interest. Differences in the rate of grassland change over time between study sites were examined by conducting ANCOVA and pair-wise comparisons (Zar 1999). Differences in the distribution of sizes of grassland patches between 1952 and the most recent year at a given study site were tested using nonparametric Kolmogorov-Smirnov (K-S) distributions tests.

Probability of Encroachment

We conducted an exploratory analysis addressing factors influencing change from grassland to woody cover from 1952 to 1992 using logistic regression. Areas of change from grassland to woody cover (shrubs or trees) (1) were contrasted with no change (0) and change from woody to grass (0). Variables considered for modeling included elevation (m), wetness (ratio slope/m^2), solar radiation (WH/m^2), Chinook winds (7 categories from west [7] to east[0]), soil coarseness (sandy loam, glacial till, luvisols and eluviated or gleyed soils), bedrock (lower Mesozoic-lower cretaceous, upper Paleozoic, lower Paleozoic), east-west gradient (UTM coordinate) and a random effect for transect ID (0-363). We evaluated 17 candidate models.

3.3 Results

Classification Validation

Overall producer accuracy of vegetation classification was 76% with the most problematic vegetation class being shrublands (Table 3.2). Conifer forest (90%) and grassland (80%) cover types were most accurately identified from aerial photographs. Grasslands were most frequently misclassified as shrubland (14%). Mixed forest areas were moderately well classified (77%). Classification of shrubland correctly classified only 58% of the time. Shrubland was most often misclassified as conifer forest (28%) or grassland (14%). Overall user classification accuracy was 80%.

Change in Extent of Cover Types

There was a relatively consistent change in vegetation types over time among sites with the exception of YHT (Fig. 3.2, Fig.3.3, Appendix V). Excluding YHT, grasslands declined on average 47.2 ± 12.1% (161 ± 195 ha) across sites between 1952 and 1992 with a corresponding increase of 38.6 ± 10.4% (145 ± 176.9 ha) in conifer forest (Fig. 3.3). WL exhibited the greatest relative loss of grassland within the upper RDRV since 1952 (73.7%, 488 ha by 2003). Grassland loss at WL was greatest between 1962 and 1982 and after 1992, Fig. 3.3). At TY there was a steady increase in coniferous forest during 1952 - 1992 (35%, 50 ha) and decrease in grassland (49.2%, 67 ha) cover (Fig. 3.3). There was a decline in grassland (34.2%, 30 ha) and increase in coniferous forest (30.4%, 36 ha) at SC. In contrast, shrublands at both TY and SC remained relatively constant or declined only slightly.

Table 3.2. Classification accuracy matrix for 2003 based on comparison with ground truth points (*n* = 94) visited in 2009 at Ya Ha Tinda and West Lakes sites, Alberta, Canada. Producer accuracy (PA) is the probability that a pixel is classified correctly, while the user's accuracy (UA) is the probability that a pixel classified as a type on a map matches to what is on the ground (Pontius and Lippitt 2006).

Vegetation class	Grassland	Shrubland	Coniferous forest	Mixed-wood Forest	Total	PA (%)
Grassland	28	5	1	1	35	80.0
Shrubland	4	15	7	0	26	57.7
Coniferous forest	0	2	18	0	20	90.0
Mixed-wood forest	1	0	2	10	13	76.9
Total	33	19	28	11	94	76.2
UA (%)	84.8	78	64.2	90.9	79.5	

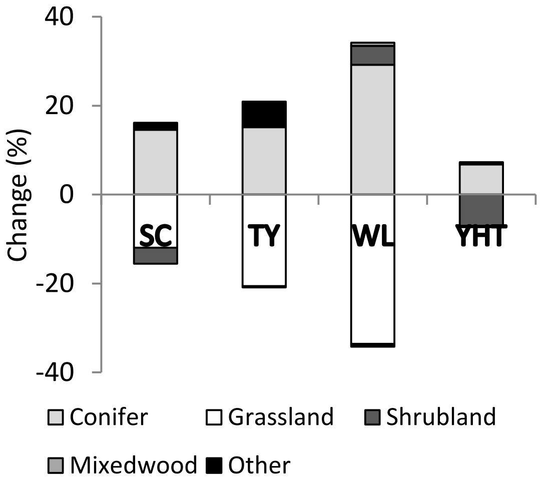


Fig. 3.2. Change (%) in vegetative cover types from 1952 - 1992 at Scotch Camp (SC) and Tyrrell Creek (TY), and from 1952 - 2003 at West Lakes (WL) and Ya Ha Tinda (YHT), upper Red Deer River Valley, Alberta, Canada.

At YHT there was an 8.8% (95 ha) decrease in grassland extent between 1952 and 1982 followed by a net increase of 9.3% (101 ha) from 1982 to 2003. During the latter period there also was a 42.9% (172 ha) decline in shrubland cover and a 9.2% (93 ha) increase in coniferous forest cover (Fig. 3.2). The pasture, which was established prior to 1952 at the YHT, comprised 0.5% (12.9 ha) of the area in the 1952 photograph but was reduced by 36.3% (8 ha) by 1972 when an established fence line became visible on the aerial photograph. Although there was an increase in coniferous forest at YHT, there also was a 2% (12 ha) removal of coniferous forest due to timber harvest in 1987 along Bighorn Creek as part of an elk habitat improvement project (Burrington et al. 1986).

Regression analysis indicated linear declines in grassland extent over the 40-year period for each site *(P* < 0.02) except at YHT. Based on ANCOVA, declines in grassland extent from 1952 to 1992 were greater at WL ($\beta = 10.93 \pm 1.54$) than at TY ($\beta = -1.63 \pm 0.15$, $P < 0.001$) and SC ($\beta = -0.73 \pm 0.17$, $P < 0.001$), whereas there was no difference in rate of grassland decline between TY and SC ($P = 0.80$).

Non-vegetated types became vegetated, and not vice versa. The largest change in a non-vegetated cover type was the development of grassland where roads or bare ground occurred formally at YHT (Appendix VI). Ponds did not occur at all study sites, and where they were found, had often become occupied by either grassland or coniferous cover by 2003.

Changes in Grassland Patch Sizes

Grassland patch distribution changed in a similar fashion at the two sites along the upper RDRV. At both SC and TY mean patch size increased, the result of a decrease in the number of smaller patches through time (Fig. 3.4). From 1952 to 2003 patch distribution at the YHT increased (K-S test: *D* = 0.15, *P* = 0.05) from a mean of 4.99 ± 45.29 (n = 215) to 14.79 ± 49.88 (n = 146), which exhibited an increase in the number of larger patches as well as a loss of smaller patches (Fig. 3.3a). Grassland patch distribution changed from 1952 to 2003 at WL (*D* = 0.27, *P* = 0.028) representing a shift from a mean of 9.33 ± 70.8 (n = 71) to 3.29 ± 7.41 (n = 53), largely driven by an increase in number of small patches in the 0.1-1 ha category (Fig. 3.2b). At TY, grassland patch size increased from 1952 to 1992 (*D* = 0.417, *P* = 0.02) from a mean of 3.77 ± 11.9 (n = 36) to 3.84 ± 7.05 (n = 18), largely driven by a reduction in the number of small patches (Fig. 3.3c). Grassland patch size distribution at SC changed from 1952 to 1992 (*D* = 0.37, *P* = 0.03), representing a shift from a mean patch size of 2.03 ± 11.5 (n = 43) to 2.29 ± 9.56 (n = 25), which accompanied the loss of patches in both larger and smaller size classes (Fig. 3.3d).

Table 3.3. Beta coefficient, least likelihood, K, AIC, and DAIC for covariates included in the top 5 logistic regression models for the progression of woody plant encroachment within the Red Deer River Valley, Alberta, Canada (1952-1992).

Model Rank	Variables	B	LL	K	AIC	ΔAIC	W
1	Soil (till)	1.25	-894	5	1798	0	1
2	Soil (till)	0.741	-991	6	1994	196	0
	Bedrock	-0.03					
3	Soil (luvisol)	0.63	-2461	6	4935	3137	0
	Chinook	0.16					
4	Soil (loam)	0.32	-2976	6	5964	4166	0
	Wetness	68.46					
5	Soil (till)	0.007	-3275	6	6562	4764	0

Probability of Encroachment

Soil was included in all top 5 models. Soil characteristics (Appendix VII) had the most weight of evidence for explaining where the transition from grassland to woody vegetation occurred within the Red Deer River Valley. In this ecosystem, well-drained coarse soil types (brunisolic, gray luvisols, and sandy loam outwash, Table 3.3) were more readily invaded than the high-nutrient, eluviated black chernozems. The vegetation of the montane ecosystem in the Red Deer River Valley may also react to moisture-related variables, such as ‘Chinook’, which incorporated differences in desiccation-prone western facing slopes compared to the east, and ‘Wetness’, which is a measure of drainage for a given unit area. ‘Chinook’ and ‘Wetness’ occurred in the third and fourth top ranked regression models, alongside ‘Soil’.

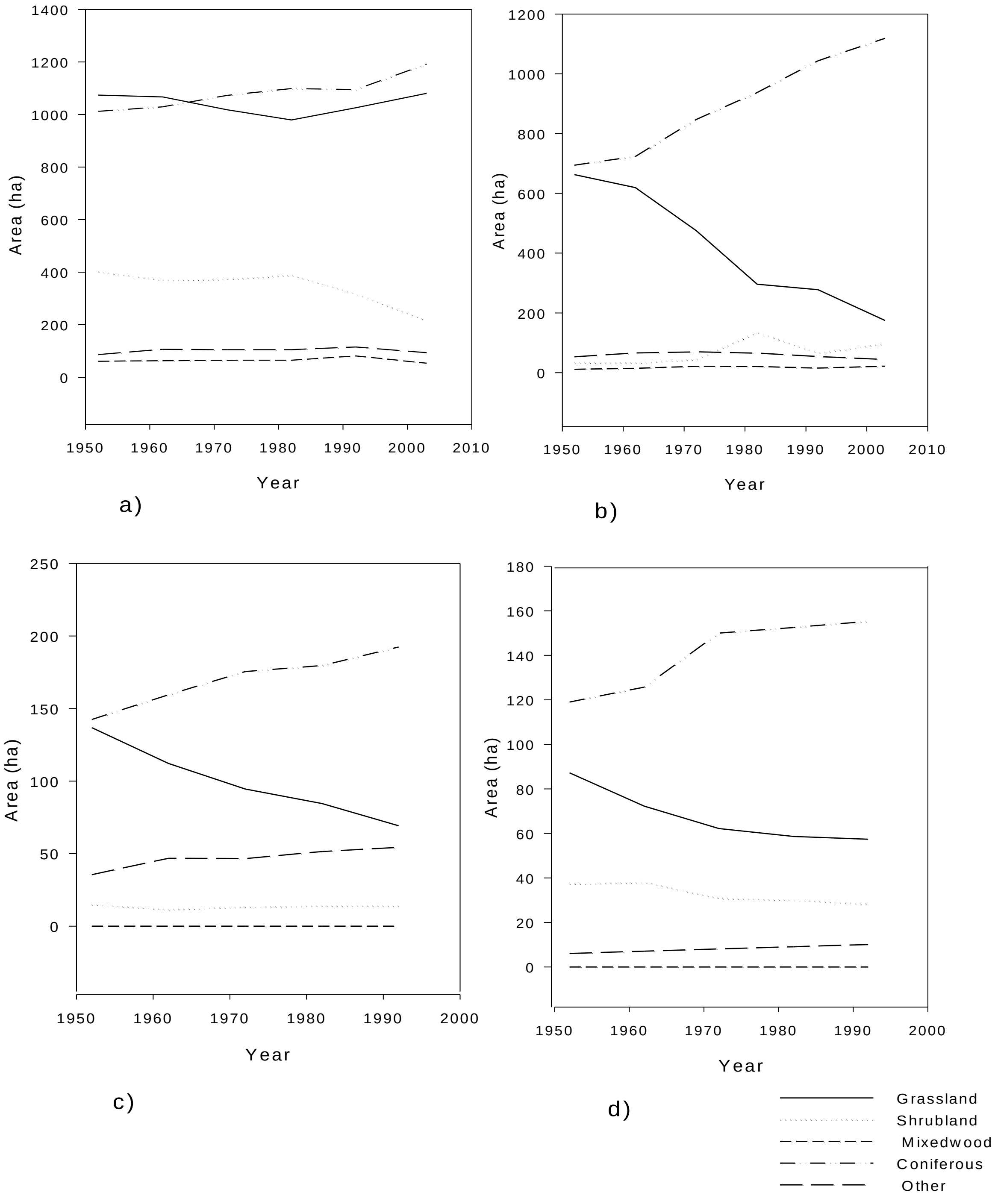


Fig. 3.3. Change in land cover within a) Ya Ha Tinda, b) West Lakes, c) Tyrrell Flats, and d) Scotch Camp, spanning a 40-yr. period (1952 - 1992) and quantified at a decadal interval. Data is from the upper Red Deer River Valley in the central east slopes of Alberta, Canada.

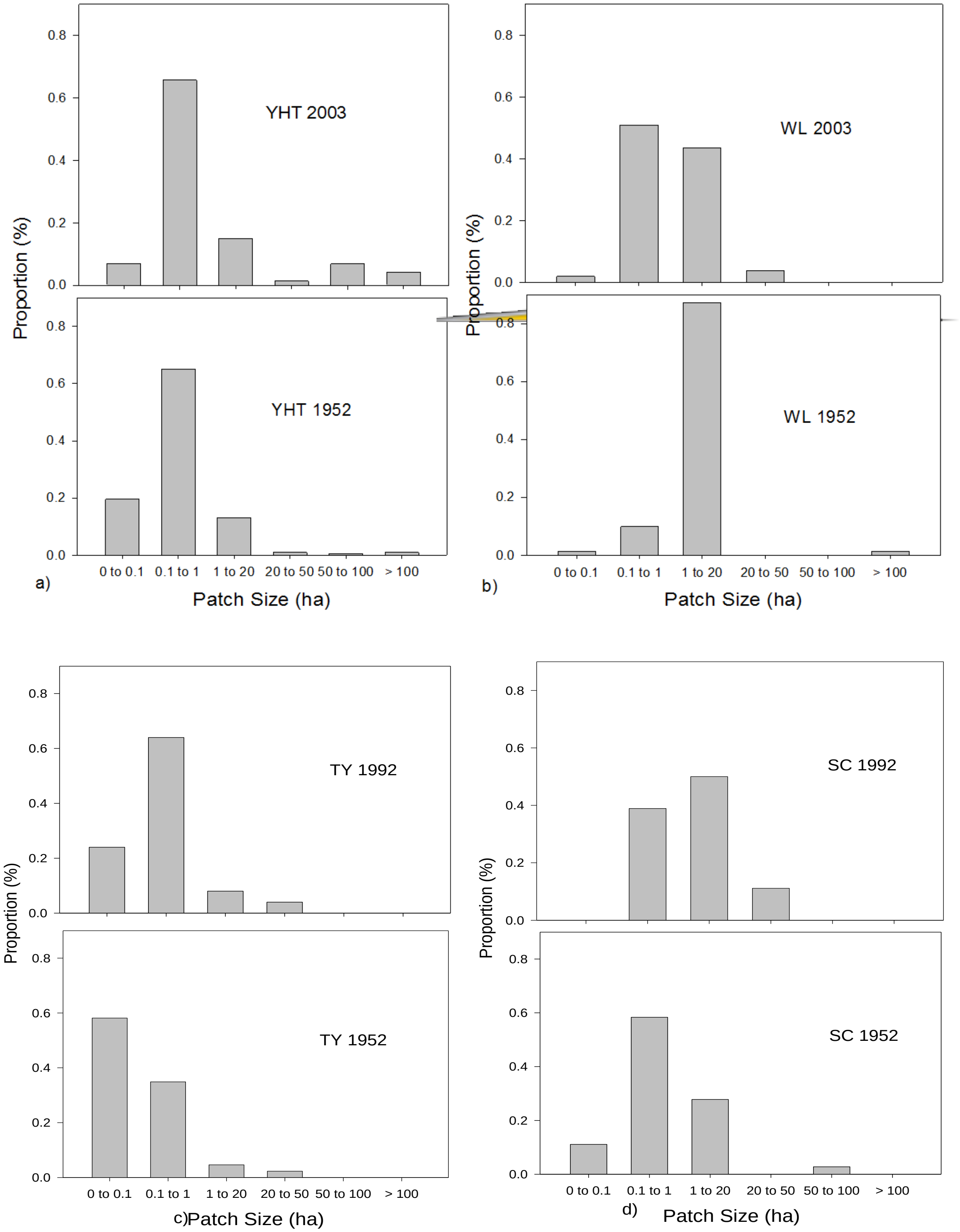


Fig. 3.4. Grassland patch distribution among the 4 study sites: a) Ya Ha Tinda (YHT) b) West Lakes (WL), c) Tyrrell Creek (TY), and d) Scotch Camp (SC) of the upper Red Deer River Valley, Alberta, Canada.

3.4 Discussion

From 1890 to 1983, Parks Canada adopted a fire suppression policy (White 1985) that has resulted in an increase in extent and mean stand age of forests (Rogeau 1994). As a result, we expected a loss of grassland extent within the RDRV over time, and an increase in woody cover such as shrubs and conifers. Such increase in coniferous forest and shrub cover has been observed during the same period in the valley environments of Jasper National Park (Rhemtulla et al. 2002) and the Rocky Mountains in the Alberta foothills (Burkinshaw and Bork 2009). In the RDRV there was an average 47% decline in grassland between 1952 and 1992 at all sites except YHT. This rate was similar to Jasper National Park where grassland area was reduced by almost 50% over an 82-year period from 1915 - 1997 (Rhemtulla et al. 2002), and to portions of the Rocky Mountain Foothills Forest Reserve, where there was a 58% loss in grassland during 1958 - 1998 (Burkinshaw and Bork 2009).

In the RDRV, the greatest grassland loss occurred at WL due to conifer encroachment. Although conifer encroachment occurred across all sites (6.8 to 29.2%), a higher rate of conifer encroachment at WL (29.2%) may have occurred because no fires have been documented in this area until recently, compared to periodic prescribed fires in the upper RDRV (Ferguson et al. 2001). Fires in the upper RDRV were maintained to create a fire guard and restore 50% of the historic fire cycle (in terms of area burned for stands of different origin years). Further, conifers in this area consist primarily of lodgepole pine. Widenmaier and Strong (2010) found lodgepole pine establishment in the Cyprus Hills was associated with warm spring temperatures and high fall (Oct –Nov) precipitation. Based on changes in grassland patch sizes, tree encroachment at WL to a large extent reflected a filling in of the grassland openings across this area, whereas smaller patches at other sites were lost as well. Rhemtulla (2002) found no change in mean patch size or number of patches from 1949 to 1997, and she argued much of the homogenization may have occurred prior to 1949 and, thus, was not detected in their aerial photograph analysis.

Because higher numbers of elk have wintered in the YHT and WL than in the upper RDRV since around the 1950s, we expected higher rates of woody plant encroachment in the YHT and WL as opposed to the SC and TY sites, due to the grazing disturbance in the grasslands. Instead we found SC and TY had lower rates of encroachment, potentially due to high dominance of willow at the SC and TY sites. Willow is highly palatable ungulate browse and due to its relatively limited distribution even moderate ungulate herbivory may arrest its growth (Tercek et al. 2010). In contrast, willow is currently a minor component of the shrub community at YHT and WL, while bog birch is the dominant shrub species in shrublands. Bog birch occurs in sites with precipitation as low as 109 - 230 mm (Blanken and Rouse 1994, Arri and Turkington 2002), and does not seem to be as moisture-sensitive as willow. We expected an increasing trend in bog birch and shrubland extent over time at YHT and WL similar to adjacent provincial lands (Burkinshaw and Bork 2009) because bog birch is not highly palatable (Toweill and Thomas 2002, Sachro et al. 2005), few fires have occurred, and grasslands can become susceptible to woody plant invasion with heavy grazing (Archer et al. 1995, Roques et al. 2001, Jurena and Archer 2003, Tremblay et al. 2006, Sasaki et al. 2008). We found shrubland increased from 1952 to 1972 at YHT but by 1982 this trend had reversed. The reversed trend was due to shrub mowing that was initiated in 1982; an annual average of 0.25 km^2 of predominantly bog birch shrubland was mowed until 2003 (Parks Canada 1987, Hebblewhite 2006).

Fig. 3.5. West Lakes, Alberta, Canada, in 1918 (top) and a repeat, spatially-referenced photo from 2008 (bottom).

4.0 Forage Availability in and along Grassland Ecotones

Woody plant encroachment into grasslands changes the understory vegetation and forage availability for ungulates. As woody cover of shrubs or trees increase, light to the understory is reduced and microclimate is modified (Jeffrey et al. 1998, Watson et al. 2003), altering understory growth, vigor and ultimately, production (Hobbs and Mooney 1986). To assess changes in forage availability with woody encroachment, we quantified species composition, richness, and biomass during peak growing season along ecotones of the 4 major cover types whose extent was delineated in Section 3.

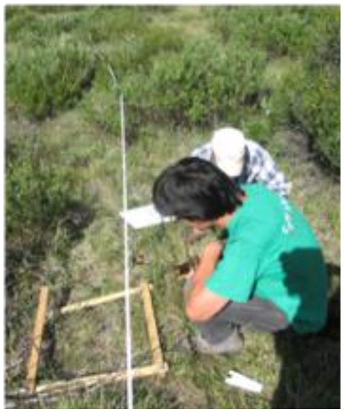
4.1 Methods

Sampling Design

We sampled ecotone vegetation between 1 July and 25 July 2009, which was considered by previous studies to be peak biomass in the area (Sachro et al. 2005, Hebblewhite et al. 2008). Sampling was first initiated at the lower elevation sites (YHT: 1-5 July; WL: 5-10 July) and extended to the higher elevation sites later in the season (TY: 14-19 July, SC: 20-25 July). The 5 types of ecotones were sampled by placing 100-m transects across an ecotone with the transect mid-point at the approximate transition point between the vegetation types; as such, a transect sampled approximately equal portion or area of each vegetation type. With the exception of the mixed-wood forest ecotone, which was limited at most sites, at least 5 transects of each ecotonal type were sampled at each site (Table 4.1).

Table 4.1. Transect types and number of times sampled within the Red Deer River Valley, Alberta, Canada, in summer, 2009. The 4 sites are Ya Ha Tinda (YHT), West Lakes (WL), Tyrell Flats (TY) and Scotch Camp (SC). Total transects across all sites = 87.

Transect type	YHT	WL	TY	SC
Grassland-shrubland	5	5	5	8
Grassland-coniferous forest	5	10	5	6
Shrubland-coniferous forest	5	5	6	7
Grassland-mixed-wood forest	5	5	0	0
Shrubland-mixed-wood forest	5	0	0	0
Total	25	25	16	21



Ground Cover

Plots of 0.25- m^2 were placed at 10-m intervals along each 100-m transect (n = 10) for visually estimating ground cover of bare ground, woody debris, rock, ground creeping woody shrubs, litter, standing dead grass, and moss (Bryohpyta). Woody debris included any tree or shrub deadfall (>2 cm) within the plot. Ground creeping shrubs included bearberry (*Arctostaphylos* *uva-ursi*) and common and mountain juniper (*Juniperus communis* and *J. horizontalis*). We defined litter as compressed, partially decomposed vegetative matter. Standing dead grass included remnants of standing graminoids from the previous growing season.

Herbaceous Biomass and Species Occurrence

Along each transect, we used a double sampling method to estimate dry biomass (g/0.25 m^2) of total graminoid and total forb species separately in each of the 10 plots. The first and last plots along each transect were clipped to within 2 cm of ground height, graminoids and forbs were separated, and material was oven dried for at least 24 hours at 100° C to derive relationships between visually estimated biomass and dry biomass by forage class. In addition to total biomass, we estimated the percent of total graminoid biomass for each grass species, and percent of total forb biomass for a subset of forb species found to be important elk forage (Table 4.2). In addition to visually estimating biomass, we recorded all graminoid and forbs species present in each of the 0.25-m^2 plot.

Table 4.2. Percent of diet of top forbs consumed by the Ya Ha Tinda elk herd based on microhistological analysis of feces collected in summer 2002 (Hebblewhite 2006).

	Summer	Winter	Current study
Asteraceae spp.	0.29	0.76	*Achillea millefolium* *Aster alpinus*
Solidago spp.	0.41	0.26	*Solidago multiradiata*
Oxytropis spp.	0.42	0.38	*Oxytropis sericea* *Oxytropis monticola* *Oxytropis cusickii* *Oxytropis splendens*
Potentilla spp.	---	1.58	*Potentilla diversifolia* *Potentilla hookeriana* *Potentilla gracilis* *Potentilla anserina*
Astragalus alpinus	3.95	1.87	*Astragalus alpinus*
Equisetum	1.21	1.70	*Equisetum arvense*
Unknown	3.63	---	
Total	9.91	3.10	

Woody Cover and Biomass

Shrub canopy intercept was recorded by species along the 100-m transect using a line intercept method (Floyd and Anderson 1987) such that cover was considered continuous with gaps of ≤10 cm. When canopy of 2 shrub species overlapped, canopy intercept of each species was recorded separately. As a result, total shrub cover exceeded 100%. Because shrub cover was recorded along the sampling transect, shrub cover over each 0.25-m^2 herbaceous plot along the transect could also be determined. Tree canopy cover was estimated using a spherical densitometer directly over each of the 0.25 m^2 herbaceous plots.

Biomass of shrub species was estimated based on stem density, basal diameter (BD), and BD-biomass relationships. Stem density was counted by species within 5 10-m^2 plot along the 100-m transect located at 10 m apart. A stem was considered an individual if it was rooted. BD of 5 stems of each species per plot were measured, averaged and used to estimate average shrub biomass/stem using BD-biomass relationships derived by Visscher et al. (2006). With the exception of small willow, biomass for a species per plot was the product of stem density and the average biomass/stem BD. For small willow (BD <0.7 mm) biomass was based on a BD-biomass relationship derived by harvesting stems at ground level within the YHT study site, oven-drying at 100° C for 72 h, and weighing to the nearest 0.01 g. Following Visscher et al. (2006), we compared several model types (linear, power, exponential) and found the best regression model for predicting biomass of small willow was fit by

$$Y = 0.0295\, x^{2.3291} \qquad (r^2 = 0.72,\ n = 48,\ P = 0.02)$$

where Y is biomass per stem (g) and x is the BD (Appendix VIII). Although several species of willow (*Salix geyeriana, S. candida, S. bebbiana, S. planifola, and S. scouleriana*) occurred in the study area, individuals were not identified reliably to species so measurements among species were pooled for this genus. Because *Juniper* spp., bog birch, bearberry, lodgepole pine and white spruce saplings (<2 m) are of low palatability (Cook 2002, Sachro 2005), they were omitted from estimates of browse availability, but their cover was estimated along the line transect.

Statistical Analysis

Analysis of variance (ANOVA) using mixed-effects linear models was used to contrast mean values of cover, biomass, richness between vegetation types and between sites in order to control for the random effect of transect. Bonferroni post-hoc tests were used for subsequent pair-wise comparisons between vegetation types using STATA 10 (StataCorp LP, College Station, TX). Pearson's correlation was used for assessing the strength of relationships between canopy cover and species abundance or biomass.

4.2 Results

Ground Cover, Tree Canopy Cover, and Understory Biomass Estimates

Tree canopy cover did not differ between mixed-wood and coniferous forest stands ($P > 0.15$), but forest canopy cover was higher than in grassland and shrubland areas ($P < 0.01$). Bare ground was higher in grasslands as compared to shrublands ($P = 0.02$), mixed-wood forest ($P < 0.01$), and coniferous forest ($P < 0.01$) forest. Woody debris cover was higher in shrublands compared to other vegetation types ($P < 0.01$), but did not differ between mixed-wood and coniferous forest stands ($P = 0.80$), which had more woody debris than grassland ($P < 0.01$) and shrubland ($P = 0.02$). Moss cover did not differ between shrublands and mixed-wood vegetation types ($P = 0.56$), but woody debris cover was greatest in coniferous > grassland > shrubland $\underset{\sim}{}$ mixed-wood forest. Dead grass (standing dead) cover was lower in forested plots (coniferous and mixed-wood) than in shrubland and grasslands ($P < 0.01$), which lacked a forest overstory.

Graminoid and forb biomass both varied by vegetation type and study site. Graminoid biomass was greatest in grasslands ($P < 0.02$) and in mixed-wood forest ($P = 0.01$) and lowest in shrubland and coniferous forest. Between sites, graminoid biomass was lower in grassland

(*P* < 0.01) and coniferous forests (*P* < 0.01) at WL than YHT (*P* < 0.01) (Fig 4.1). Forb biomass differed among vegetation types across study sites (*P* < 0.01), with forbs in grassland = shrubland > mixed-wood > coniferous forest. Between sites, forb biomass was similar in all vegetation types except grasslands, where the YHT was higher than WL (*P* < 0.01, Table 4.3). Shrub biomass was lowest in grasslands (*P* < 0.02) and greatest in shrublands (*P* < 0.01). Shrub biomass was not different within shrublands across sites (*P* = 0.6). Shrub biomass was higher at WL compared to YHT sites only in mixed-wood forest (*P* < 0.02), which comprised only 2% of the vegetation extent at both the YHT or WL site and was not present at the SC or TY.

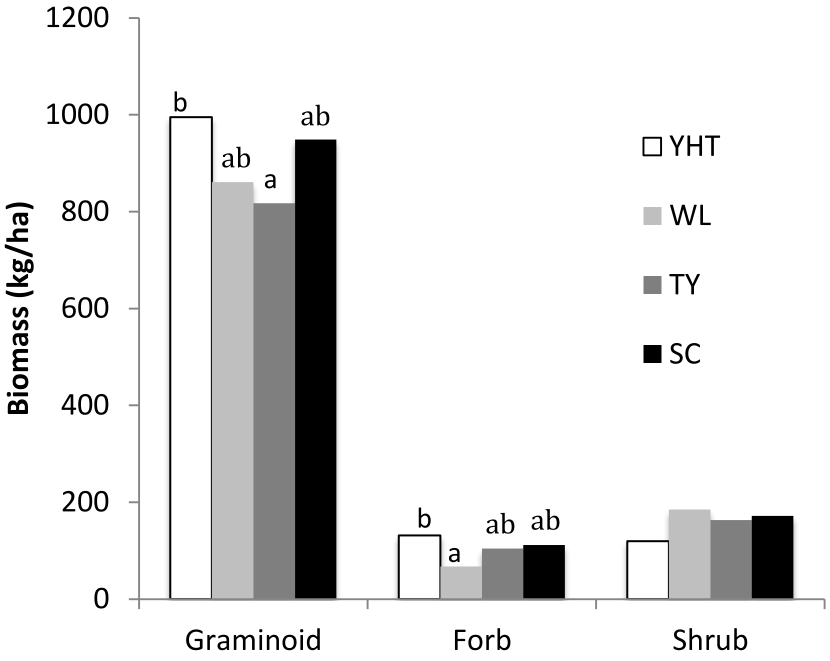


Fig. 4.1 Mean biomass (± SD) within grassland study sites (YHT, WL, TY, and SC) along the Red Deer River Valley, Alberta, Canada. Letters indicate differences between sites within plant groups (α = 0.05).

Table 4.3 Mean and standard deviation (STD) of ground cover (%) and total biomass (kg/ha) by vegetation type along ecotone transects, Red Deer River Valley, Alberta, Canada. Statistical differences indicated by *a, b, c*…etc.

	Grassland (n=304)		Shrubland (n=238)		Mixed-wood (n=80)		Conifer (n=248)	
	Mean	STD	Mean	STD	Mean	STD	Mean	STD
% Cover								
Tree canopy	0.2^{a}	3.4	0.5^{a}	5.5	60.9^{b}	17.7	57.4^{b}	19.0
Bare ground	6.0^{b}	11.9	3.4^{a}	8.7	1.0^{a}	5.5	2.1^{a}	10.1
Rock	1.1	4.3	0.7	4.8	0.0	0.0	0.7	3.4
Woody debris	6.6^{a}	15.4	32.6^{c}	28.6	17.4^{b}	16.8	15.2^{b}	18.3
Litter	28.8	19.2	27.5	17.1	35.1	22.8	43.4	27.8
Moss	13.4^{b}	19.9	7.3^{a}	12.8	3.9^{a}	13.3	21.0^{c}	26.7
Dead grass	6.7^{b}	9.7	3.4^{b}	7.1	0.8^{a}	3.3	0.6^{a}	2.9
Total	62.7	25.4	74.9	22.3	58.2	26.4	83.1	21.3
Biomass of CAG (kg/ha)								
Graminoid	880.0^{c}	530.5	593.2^{b}	494.4	735.3^{c}	562.5	164.0^{a}	183.3
Forb	99.5^{bc}	83.5	106.5^{c}	108.0	58.4^{b}	67.6	28.3^{a}	45.4
Shrub	$\underline{260.7}^{a}$	96.0	$\underline{1460.6}^{c}$	589.0	$\underline{795.2}^{b}$	611.9	$\underline{446.9}^{b}$	388.9
Total	1240.1	710.0	2160.4	1191.3	1588.9	1241.9	639.1	617.6

Species Composition and Richness

Composition of graminoid species based on visual estimates of percent of total biomass varied among communities (Table 4.4). With the exception of grassland, hairy wild rye (*Elymus innovatus*) was the most abundant graminoid species across vegetation types (ANOVA, $df = 3$, 87, $F = 21.3$, $P = 0.04$), with no difference in percent biomass between conifer and mixed-wood forests ($P = 0.14$), and mixed-wood forests and shrublands ($P = 0.06$). Rough fescue (*Festuca campestris*) was the most prominent species in grassland communities across sites (Table 4.4).

Based on percent frequency of occurrence of species within 0.25-m^2 plots, old man's whiskers (*Geum triflorum)* was the most common forb in grasslands across sites. Yarrow (*Achillea millefolium*) was common in shrubland and mixed-wood forest areas (ANOVA, $df= 3$, 87 $F = 7.3$, $P < 0.001$), and sweet vetch (*Hedysarum alpinum*) did not differ between types ($P = 0.69$). Although shrub biomass did not vary among vegetation types ($P = 0.16$), composition of shrubs did. Shrubby cinquefoil (*Potentilla fruticosa*) dominated the shrub component of the grassland vegetation type (Table 4.4). Willow was the most abundant palatable shrub in shrublands (ANOVA, $P < 0.01$), and buffaloberry (*Shepherdia canadensis*) was most abundant in mixed-wood and coniferous forests ($P = 0.01$). Bog birch, which was not considered a major browse species, had a higher proportion of shrub cover at YHT and WL, than at SC and TY, which had a larger *Salix* component. Bog birch and *Salix* spp. both dominated shrub areas, while other shrubs were less frequent (Fig. 4.2).

Ecotonal Shifts in Species Composition

A prominent shift in biomass of graminoid species occurred along ecotones (Fig. 4.3a-c). Most evident was the decrease in *Festuca campestris* and increase in *Elymus innovatus* with increasing canopy as grassland transitioned into shrubland and conifer forest. The transition was more abrupt from grassland to coniferous forests or shrublands than from shrublands to coniferous communities. Total biomass of rough fescue decreased from early to later seral stages; however, hairy wild rye was relatively constant along grassland to shrubland transitions, but increased along the transition from grassland to shrubland and shrubland to conifer.

Shifts in percent frequency of forbs varied by species. For example, there was a decline in percent frequency of old man's whiskers (*Geum triflorum*) from grassland to shrubland that was related to total shrub cover ($r = -0.88$, $P < 0.01$) (Fig. 4.4b). A similar decline occurred in the shrubland to coniferous forest in common yarrow (*Achillea millefolium*) with development of a tree canopy ($r = -0.88$, $P < 0.01$). In contrast, percent occurrence of sweet vetch (*Hedysarum alpinum*) was highly variable (Fig. 4.4c).

In terms of available shrub biomass, as grassland shifted into shrubland, there was a drop in shrubby cinquefoil and little response in willow biomass at the transition in plots 4 and 5 (Fig. 4.5.b). As a tree canopy developed there was a decrease in shrubby cinquefoil ($r = -0.6$, $P < 0.01$) and willow ($r = -0.4$, $P < 0.01$), and an increase in the prominence of buffaloberry ($r = 0.71$, $P < 0.01$; Fig. 4.5 a, c).

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Table 4.4. Top 5 most abundant graminoid and shrub species based on percent of estimated biomass and the most common forbs based on percent frequency of occurrence along transects in the 4 major vegetation types within the Red Deer River Valley, Alberta, Canada. Forbs are not restricted to those listed in Table 4.2 but include all forbs found in a plot.

Grassland	Mean (% biomass)	STD	Shrubland -	Mean (% biomass)	STD	Mixed-wood forest -	Mean (% biomass)	STD	Coniferous forest -	Mean (% biomass)	STD
Graminoids											
Festuca campestris	41.2	32.0	*Elymus innovatus*	44.0	36.8	*Elymus innovatus*	55.9	43.0	*Elymus innovatus*	66.2	42.3
Elymus innovatus	21.9	23.8	*Festuca campestris*	23.5	30.2	*Festuca campestris*	10.8	18.8	*Bromus inermis*	8.6	24.4
Carex spp.	11.5	20.6	*Carex* spp.	10.1	19.3	*Carex* spp.	8.9	25.4	*Rough fescue*	7.8	19.7
Bromus ciliates	5.4	14.1	*Bromus inermis*	7.8	18.7	*Muhlenbergia richardsonis*	4.2	16.8	*Carex spp.*	5.0	18.1
Helictotrichon hookeri	2.9	8.5	*Juncus balticus*	3.7	12.4	*Agropyron trachycaulum*	1.4	6.7	*Muhlenbergia richardsonis*	2.2	12.3
Forbs	Mean (% occurrence)	STD	-	Mean (% occurrence)	STD	-	Mean (% occurrence)	STD	-	Mean (% occurrence)	STD
Geum triflorum	0.5	0.5	*Achillea millefolium*	0.4	0.5	*Achillea millefolium*	0.4	0.5	*Hedysarum alpinum*	0.3	0.5
Astragalus alpinus	0.4	0.5	*Solidago multiradiata*	0.3	0.5	*Galium boreale*	0.4	0.5	*Solidago multiradiata*	0.3	0.4
Galium boreale	0.4	0.5	*Galium boreale*	0.3	0.5	*Vicia americana*	0.3	0.5	*Astragalus alpinus*	0.2	0.4
Anemone multifida	0.4	0.5	*Fragaria viriginiana*	0.3	0.5	*Hedysarum alpinum*	0.3	0.4	*Fragaria virginiana*	0.2	0.4
Solidago multiradiata	0.4	0.5	*Hedysarum alpinum*	0.3	0.5	*Fragaria virginiana*	0.2	0.4	*Achillea millefolium*	0.2	0.4
Shrubs	Mean (% biomass)	STD	-	Mean (% biomass)	STD	-	Mean (% biomass)	STD	-	Mean (% biomass)	STD
Potentilla fruticosa	0.9	1.0	*Potentilla fruticosa*	0.1	0.9	*Potentilla fruticosa*	0.4	0.4	*Shepherdia canadensis*	0.8	0.9
Salix spp.	0.1	0.0	*Salix* spp.	0.6	0.0	*Shepherdia canadensis*	0.4	0.4	*Potentilla fruticosa*	0.1	0.1
			Shepherdia canadensis	0.1	0.0	*Populus tremuloides*	0.2	0.1	*Salix* spp.	0.1	0.0
			Populus tremuloides	0.1	0.0						
			Rosa acicularis	0.1	0.0						

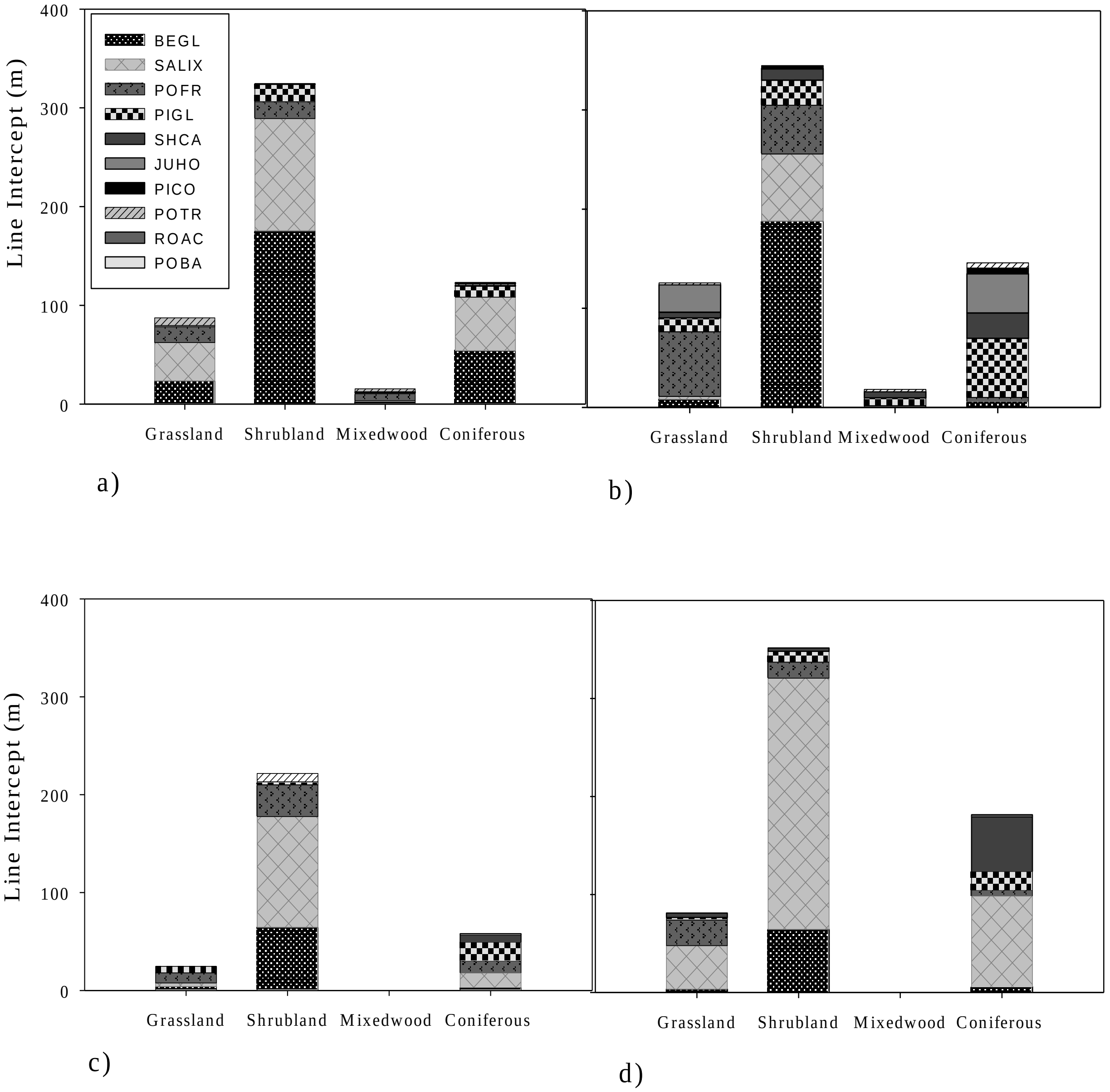

Fig. 4.2. Shrub dominance within vegetation types at a) YHT, b) WL, c) TY, and d) SC study sites, upper Red Deer River Valley, Alberta, Canada. Shrub dominance was measured using the line intercept method, along ecotone transects. Units are comparable within vegetation types. BEGL (*Betula glandulosa*) JUHO (*Juniperus horizontalis*), PICO (*Pinus contorta*) PIGL (*Picea glauca*) POFR (*Potentilla fruticosa*) POTR (*Populus tremuloides*) SHCA (*Shepherdia canadensis*) ROAC (*Rosa acicularis*) SALIX (*Salix* spp.) POBA (*Populus balsamifera*).

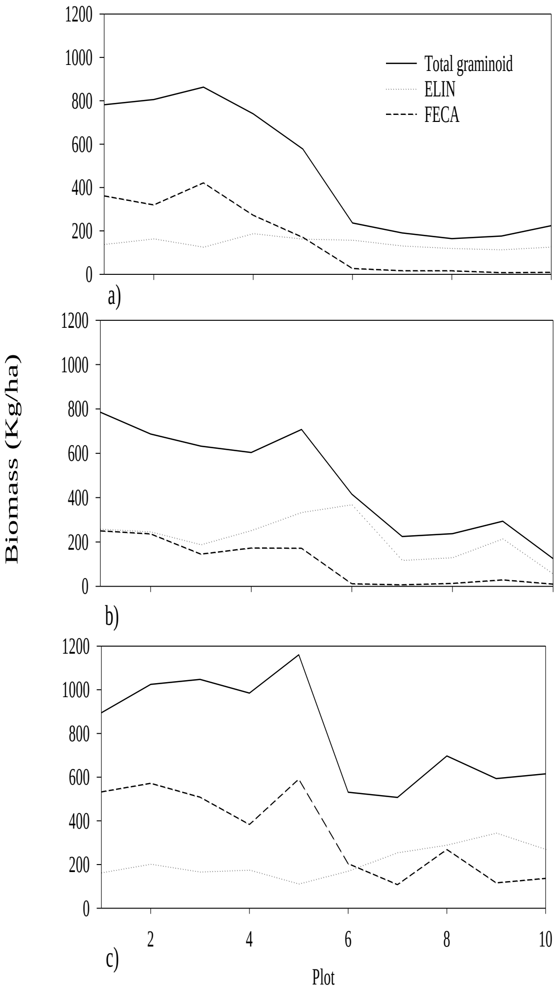

Fig. 4.3. Biomass (kg/ha) of all graminoids, hairy wild rye (*Elymus innovatus*, ELIN), and rough fescue (*Festuca campestris*, FECA) across a) grassland to coniferous forest (n=30), b) grassland to shrubland (n= 30) and c) shrubland to coniferous forest (n= 22) ecotones within the Red Deer River Valley, Alberta, Canada.

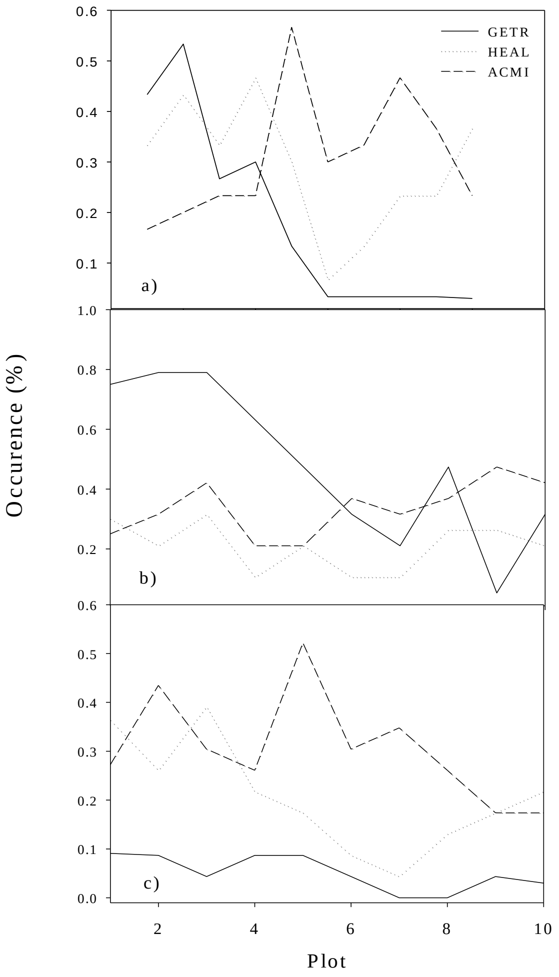

Fig. 4.4 Frequency of occurrence (%) of prominent forb species, across a) grassland to coniferous, b) grassland to shrubland and c) shrubland to coniferous ecotones within the Red Deer River Valley, Alberta, Canada. HEAL (*Hedysarum alpinum)*, ACMI (*Achillea millefolium*), GETR (*Geum triflorum*).

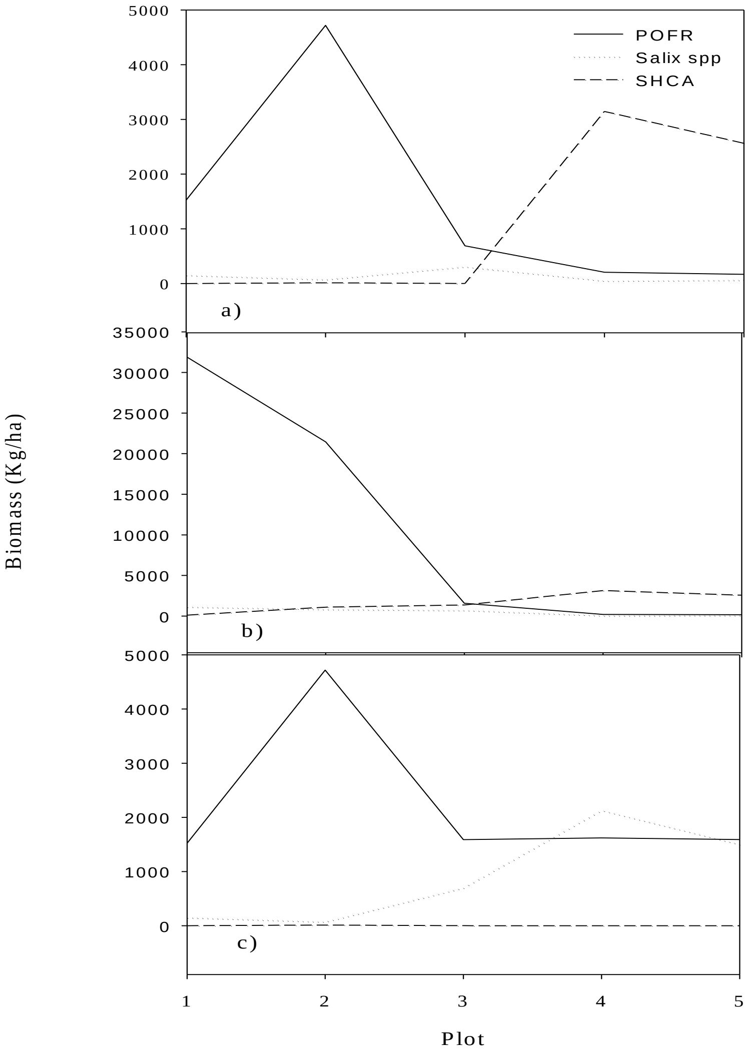

Fig. 4.5 Biomass (kg/ha) for the most dominant shrub species, *Potentilla fruticosa* (POFR), *Shepherdia canadensis* (SHCA), and *Salix* spp., averaged across 4 sites within the Red Deer River Valley, Alberta, Canada. Plots spanned 100-m transects along a) grassland to coniferous, b) grassland to shrubland and c) shrubland to coniferous forest ecotones.

Overstory-Understory Relationships

Because understory production differed between forest communities (*see* above), we derived understory-overstory relationships by vegetation community. In forested communities, herbaceous understory biomass was negatively related to both tree and shrub overstory with graminoids being more influenced by overstory cover in mixed-wood forest than in coniferous forests (Table 4.5, Fig. 4.6). There was no relationship between shrub cover and tree cover in either forest community ($P > 0.29$). In shrubland communities, there was no difference in understory production among sites; thus, one relationship was developed between shrub cover and understory biomass. Shrub cover had a stronger negative relationship on graminoids than forbs (Table 4.5, Fig. 4.7). Exponential relationships did not explain a more substantive amount of variation (<3%) than simple linear relationships.

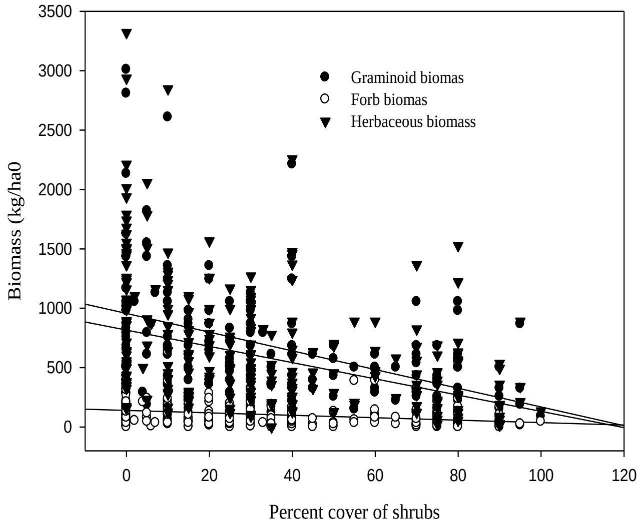

Fig. 4.6. Relationships between overstory shrubs (%) and understory graminoids, forbs and herbaceous biomass (kg/ha) in shrubland transects across 4 sites in July 2009, Red Deer River Valley, Alberta, Canada.

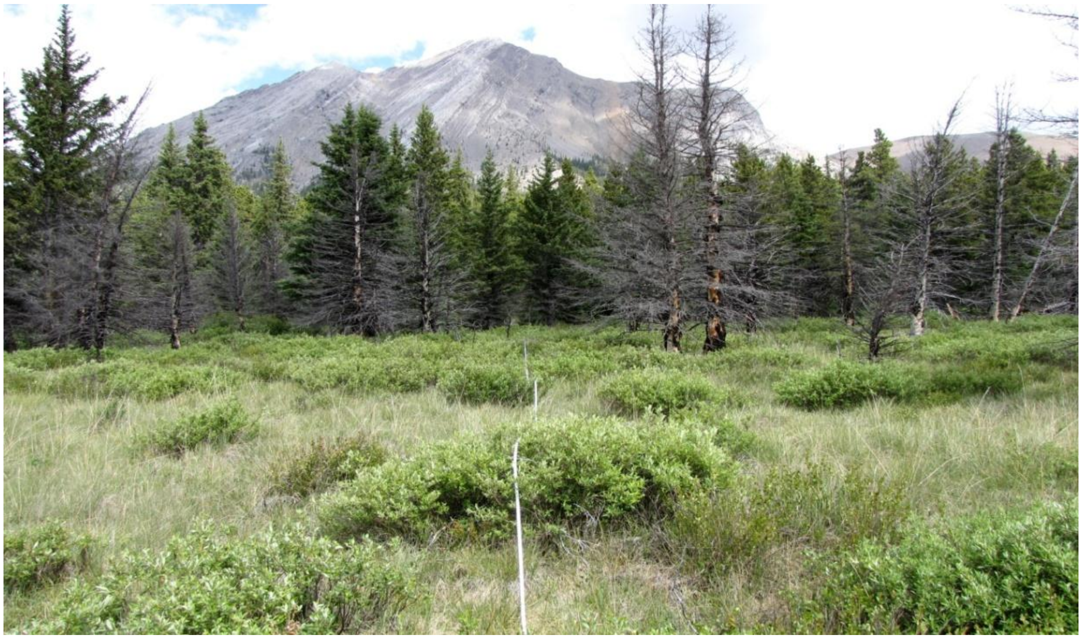

Table 4.5. Linear relationships between overstory cover (%) and understory green biomass (kg/ha) measured within 0.25-m^2 plots along transects at 4 sites in the Red Deer River Valley, Alberta, Canada, in July 2009. Presented are β coefficients, robust standard errors adjusted for transect, and *P*-values indicating significance levels of the β coefficients.

	Graminoids (kg/ha)			Forbs (kg/ha)			Herbaceous (kg/ha)		
	β	S.E.	*P*	β	S.E.	*P*	β	S.E.	*P*
Mixed wood forest									
Tree cover	-9.8	3.5	0.01	-0.76	0.36	0.05	-10.5	3.63	0.01
Shrub cover	-10.2	2.6	<0.01	-1.16	0.67	0.10	-11.4	3.10	<0.01
Intercept	1505.3	305.4	0.01	119.7	35.3	<0.01	1625.0	328.2	<0.01
n = 79			$r^2 = 0.17$			$r^2 = 0.08$			$r^2 = 0.18$
Coniferous forest									
Tree cover	-1.6	0.83	0.05	-0.53	0.21	0.02	-2.22	1.01	0.03
Shrub cover	-1.7	0.45	<0.01	-0.24	0.11	0.04	-1.99	0.51	<0.01
Intercept	287.4	57.42	<0.01	62.7	16.6	<0.01	350.1	70.3	<0.01
n = 248			$r^2 = 0.07$			$r^2 = 0.06$			$r^2 = 0.08$
Shrubland									
Shrub cover	-6.8	1.23	<0.01	-1.0	0.24	<0.01	-7.9	1.34	<0.01
Intercept	816.0	83.7	<0.01	140.9	13.5	<0.01	956.1	90.6	<0.01
n = 238			$r^2 = 0.16$			$r^2 = 0.07$			$r^2 = 0.17$

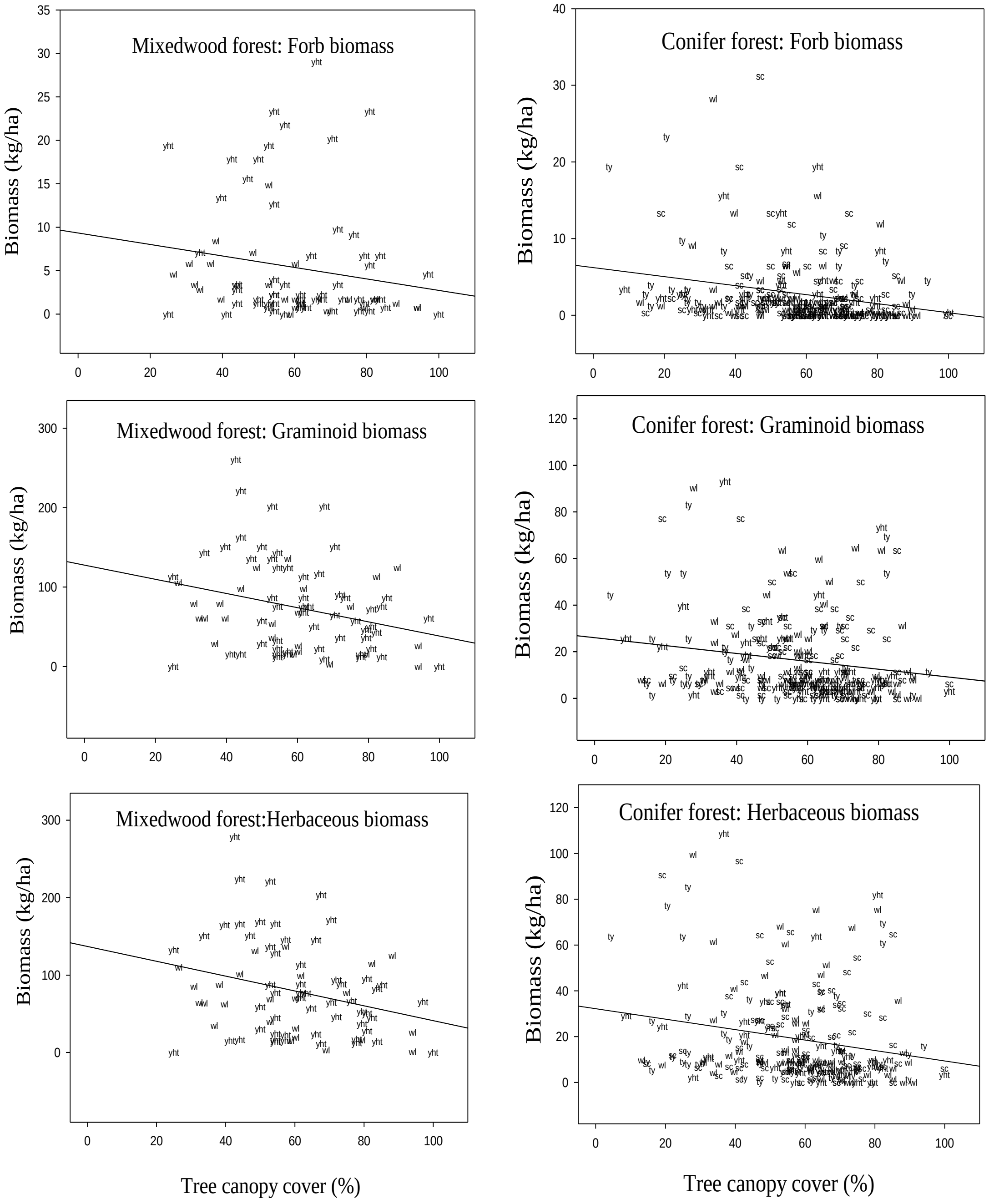

Fig. 4.7 Influence of overstory tree canopy cover on total herbaceous (forb plus graminoid), graminoid, and forb biomass (kg/ha) in the mixed-wood and coniferous forest type at the YHT, WL, TY and SC study sites, Alberta, Canada.

4.3 Discussion

The objectives of the current study were to assess changes in biomass and composition of herbaceous and woody species between vegetation types and across ecotones in the upper RDRV. Current year annual growth of graminoids (995 kg/ha), forbs (131 kg/ha), and shrubs (120 kg/ha) in grassland plots at YHT in July 2009 were within the range of annual estimates reported for graminoids (735 to 1018 kg/ha), forbs (250 to 403 kg/ha) and shrubs (41to 395 kg/ha) in section 5 of this report, but considerably higher than the total CAG reported by McInenly (2003: 297 kg/ha) outside 6 exclosures on the YHT grassland. Across the broader area of eastern Banff NP and the RDRV, Hebblewhite (2006: Appendix A2.5a), incorporating data from Sachro (2005), also reported generally lower graminoid (364 kg/ha), forb (230 kg/ha), and shrub (137 kg/ha) biomass in open herbaceous sites during 2001 - 2003, particularly given their measures reflect total standing biomass rather than CAG. The differences mostly likely reflect the drought conditions that occurred during these years. Somewhat higher production than 2003 was also reported in 2005 in the controls of a burn study in alpine grasslands in Banff (Tannas et al. 2006), but the differences over time were not as great likely due to a later growing season in these sites. Further, our estimates of total herbaceous production for SC are more similar to those reported for the unburned sites in 2003 (892 kg/ha) than in 2001 (456 kg/ha). Differences were attributed to reduced grazing intensity by horses and elk (Eastern Slopes Rangeland Seeds Ltd 2003).

Production in terms of CAG in grasslands at YHT was similar to that at SC and TY, located ~10 km up the Red Deer River, but was higher than at WL, even though that area was adjacent. We conclude that soils, rather than differences in climate, are likely responsible. Willoughby (2001) described two rough fescue-dominated community types within the Ya Ha Tinda ranch proper. One is a rough fescue-sedge-Junegrass (*Festuca scabrella-Carex filifolia-Koeleria macrantha*) type occurring on steeper, morainal, and colluvial slopes such as those found primarily in the WL region. This community type is found on drier sites than the rough fescue-fringed brome-sedge (*Festuca scabrella-Bromus ciliatus-Carex* spp.) community type of the YHT, where forage production tends to be higher (Willoughby 2001). In addition, the YHT area contains a higher proportion of black chernozem soils commonly known to be of high productivity (see McGillis 1976 in Morgantini 1995, Krupenikov et al. 2011). In the WL area, soils are mostly represented by the less developed and thin Orthic Regosols.

Outside of grasslands, both Hebblewhite (2006) and Sachro (2005) reported similar patterns in herbaceous understory biomass among other vegetation types. As in this study, both researchers reported the lowest total herbaceous biomass occurred in coniferous sites (< 325 kg/ha) with a similar decline in understory as canopy closure increased (Hebblewhite 2006: A2.5a). In contrast to Hebblewhite (2006), we found reduced (~33%) graminoid biomass in shrublands, that was related to the amount of shrub overstory (Fig 4.2). Differences among studies are may be due to the high variation in shrub abundance across sites. Even though shrub biomass was highest in both shrublands and mixed-wood forests across studies, comparisons in shrub biomass values among studies are complicated by differences in methodologies.

Results from the combined vegetation studies in the upper RDRV have important implications for changing availability of forages to ungulates. Because grassland and shrublands provide the most abundant forage for ungulates, long-term conifer encroachment since the 1950s indicates reduced carrying capacity for ungulates in the vicinity of these grassland systems. Winter forage loss with conifer encroachment is exacerbated by a shift away from the predominance of rough fescue, whose winter-hardy structure and high palatability makes it important forage. Periodic natural or prescribed burning by Parks Canada around the grasslands at SC or TY has not been sufficient to alter the long-term trajectory of tree encroachment based on our photo-interpretation studies. This is consistent with the analysis of White (2001:67) who reported that fire history patterns around meadows did not strongly support the hypothesis that meadows were a focus for frequent cultural burns. However, short-term improvements in forages from prescribed burning by Parks Canada may temporarily offset some grassland losses. For example, the SC grasslands were burned in 2000, and improved forage production occurred in 2001 (65 kg/ha increase), but the difference was gone by 2003 (Eastern Slopes Rangeland Seeds Ltd 2003). A major improvement from burning may have been the removal of standing dead, which is indicated by a large increase (27%) in litter on burned sites from 2001 to 2003.

Although total biomass in shrublands did not vary among sites, the quality of shrub biomass differed greatly. Because bog birch was more abundant at WL and YHT compared to TY and SC where willow was most abundant, shrub encroachment at YHT and WL has provided little new browse while reducing the availability of herbaceous understory. At YHT, mowing of shrub stands to enhance winter range started in the mid-1980s (YHT Management Plan 1987, Seel and Wiebe 1988) and continued through the early 2000s (E. Merrill, personal observation). Initially, mowing reduced shrubs including willow. Due to high elk populations (800 – 2,000 during winter) over the past ~15 years (Hebblewhite et al. 2006), it is likely that elk herbivory has altered shrub recovery to at least some degree as a result of selective grazing, as occurred with aspen (White et al. 1998). Willow is preferred elk forage particularly in late summer while bog birch is not (Sachro et al. 2005, Hebblewhite 2006, van Beest et al. 2010). With the recent (post-2005) reductions in elk densities, we found some evidence for the resurgence of willow at the YHT. Densities of willow shoot stems <0.7 mm diameter were greater than 97 shoots/m^2. Whether these plants have escaped herbivory because of the low elk population remains to be seen.

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5. Long-term Monitoring of Vegetation

To document annual vegetative growth over the season and end-of-summer forage abundance on the YHT grassland, we continued to monitor vegetation on the YHT grassland from 2008 - 2010 and compared these results to earlier years when vegetation sampling was similar.

5.1 Methods

We used the spatial extent of the YHT grassland digitized by McInenly (2003) from a satellite image (excluding continuous forest portions), and verified in the field as non-forest based upon canopy cover of <60%. Vegetation was sampled at 45 locations that were at least 500 m apart on a sampling grid (Fig. 5.1). In most cases vegetation was sampled monthly between 2008 and 2010. Total standing biomass (TSB), including new growth (CAG) and standing last year's growth (LYG) was determined using a "disc-pasture-meter" measurement (Vartha and Matches 1977, Dorgeloh 2002) in each of the 4 corners of a 5x5-m^2 plot at each of the 45 sites. This method measures plant "bulk density" and consists of sliding a 0.25-m^2 base plate (weight = 222 g) over a 1-m long calibrated aluminum meter stick. After dropping the disc from the top of the meter stick, the settling height (accurate to 0.5 cm) was recorded. In every plot, the proportion of TSB that was CAG and LYG was visually estimated, and cover was visually estimated for the broad vegetation classes of grasses, forbs, shrubs, and bare ground. Shrubs primarily included *Potentilla* spp.

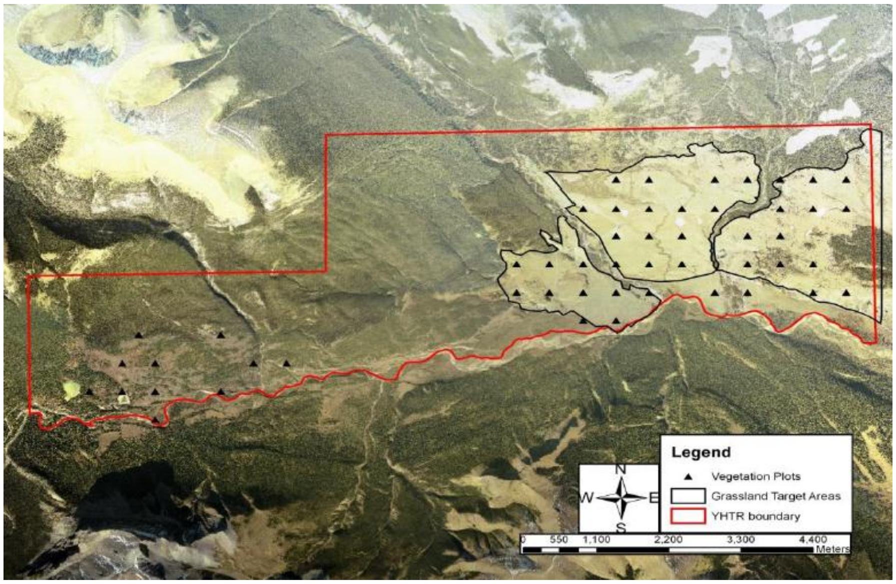


Fig. 5.1. Location of 45 permanent grassland sites where vegetation was sampled from 2005 - 2010. Sites are part of a more extensive grid randomly located on the Ya Ha Tinda ranch, Alberta, Canada, for purpose of counting elk pellet groups. Figure credit: H. Spaedtke (2009).

Height of the drop disc method was related to total standing biomass (TSB) by clipping all TSB in 282 plots during 2005 - 2007, directly adjacent to the permanent vegetation plots. Once a drop disc measurement was taken, TSB greater than 2 cm in height was clipped, dried at 100^{o} for at least 24 hours and weighed. A linear regression of biomass (g/0.25 m^2) dry weight and drop disc height was developed that explained 76% ($P < 0.001$) of the variation in dry weight, and multiplied by 4 to report plant biomass on a m^2 basis (Spaedtke 2009).

Monthly estimates of CAG within a plot were derived as the product of the TSB (derived from the recorded height of the drop disk) and the visually estimated proportion of TSB that was current growth (CAG). CAG attributed to forbs, graminoids, and shrubs was determined from the product of CAG and the relative proportion of cover of each plant group, where relative proportion denotes the proportion of vegetative cover within a plot. At each site, the vegetation values of the 4 plots were averaged, and we report values that represent monthly means ± SE across sites (n = ~45/sampling period). For comparison, we calculated similar measures for the same sites in 2005 - 2007 based on data collected by Spaedtke (2009).

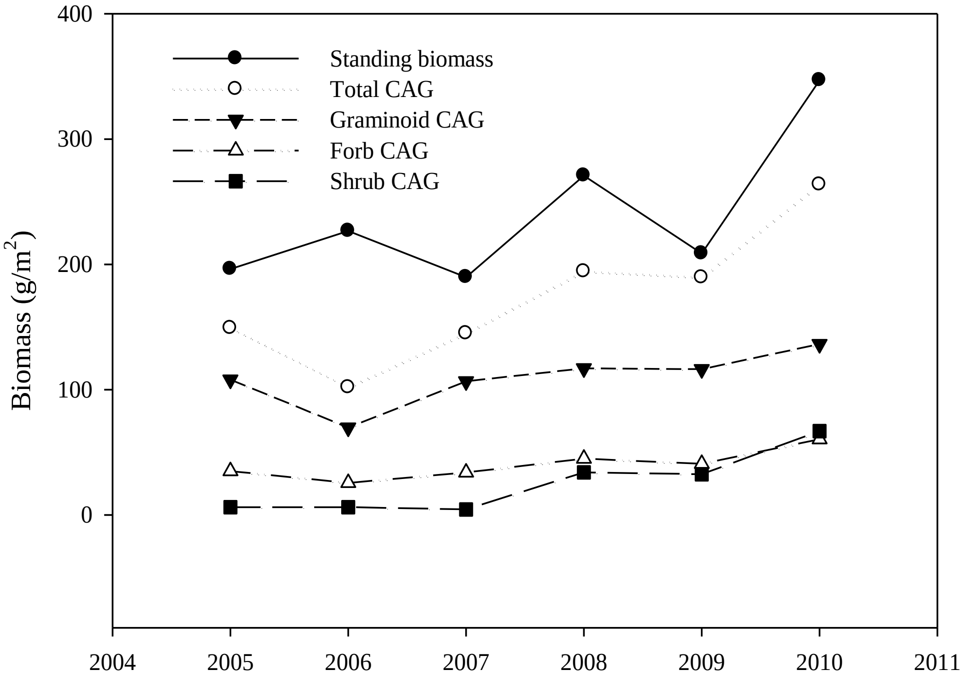


Fig. 5.2. Standing plant mean biomass and current annual growth (CAG) in g/m^2 on the Ya Ha Tinda grassland, Alberta, Canada, August 2005-2010.

5.2 Results and Discussion

Current annual growth (CAG) appears to peak in August at YHT (Table 5.1). Total CAG in August ranged between 101 - 264 g/m^2 with forbs comprising 22 - 25% and shrubs 4 - 6% of the total CAG in the grasslands (Table 5.1). These estimates are higher than those reported by McInenly (2002) for sites outside 6 exclosures at YHT in June 2002 during a drought, but are comparable to results reported by Merrill et al. (2007) from the same 6 exclosures in July 2007. Spaedtke (2009) reported that the mean standing biomass (CAG plus previous annual growth) measured during summer in the years before aversive conditioning (2001 – 2004) was less than half the value (87.14 g/m^2 ± 9.13 SE) than in the years (2005 - 2007) with (183.31 g/m2 ± 5.79 SE). He attributed the difference, in part, to an approximately 45% increase in mean precipitation that occurred in the years before aversive conditioning (Fig. 5.3).

Since 2007, CAG has fluctuated among years but with a slow increase (Fig. 5.2). Reduced summer grazing due to aversive conditioning (Spaedtke 2009) or reduced elk population may be related to the increase in graminoids. For example, Merrill et al. (2007) reported higher biomass and an increase in the basal diameter of fescue plants inside exclosures at YHT, resulting from exclusion from grazing. The drop in total standing biomass in 2009 was likely related to the burns in the grasslands.

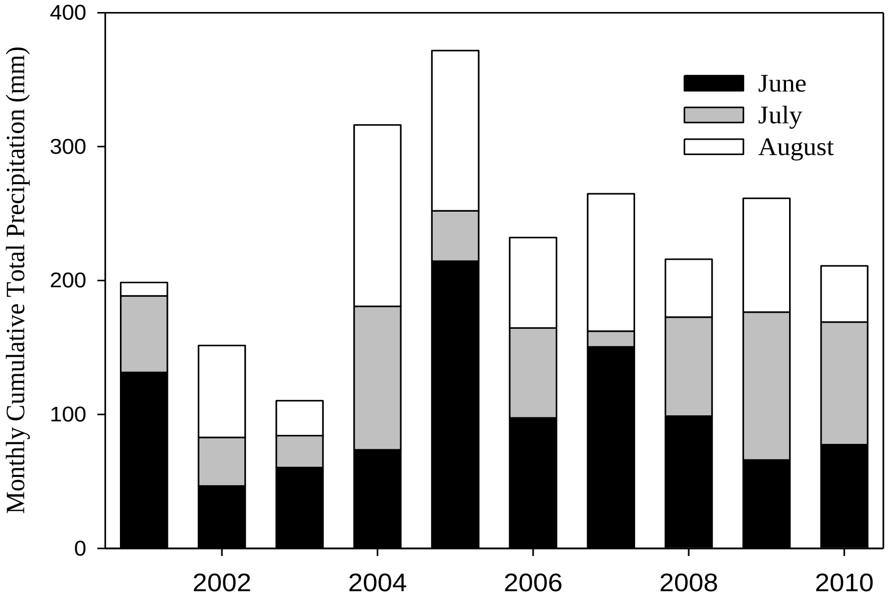


Fig. 5.3. Total cumulative precipitation June - August of 2001 - 2010, recorded at Blue Hill weather station, Alberta, Canada. July 2010 data were missing; an adjusted value based on data from Scalp Creek in 2009 and 2010 is presented.

Table 5.1. Mean ± SD of total standing biomass (standing dead plus CAG), total current annual growth (CAG), and current annual growth of graminoids, forbs, and shrubs (g/m^2) in the grassland of YHT ranch, Alberta, Canada, 2005 - 2010.

	Standing biomass		Total CAG		Graminoid		Forb		Shrub		CAG
	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD	(%)
May											
2005	94.9	45.0	35.1	16.6	27.9	13.2	6.0	2.8	1.2	0.6	0.37
2006	75.7	26.4	24.2	10.7	19.5	9.4	3.9	2.9	0.8	2.1	0.32
2007	61.6	23.9	21.1	8.7	16.5	7.1	4.0	3.2	0.6	2.0	0.34
2008	161.8	229.5	43.8	50.9	22.6	21.2	8.9	11.9	12.3	30.0	0.27
2009	161.6	152.8	43.2	47.8	26.1	26.1	8.3	13.5	9.1	16.0	0.27
2010	--	--	--	--	--	--	--	--	--	--	
June											
2005	146.4	32.5	90.0	20.0	63.4	14.1	24.0	5.3	2.6	0.6	0.62
2006	142.7	40.5	82.8	30.0	58.8	24.9	20.6	11.7	3.4	6.6	0.58
2007	128.4	34.4	84.0	31.0	59.3	25.2	23.5	15.8	1.2	2.0	0.65
2008	234.6	169.4	120.5	67.0	68.0	38.9	27.8	20.8	24.6	43.8	0.51
2009	163.1	107.1	71.3	42.6	47.2	28.2	16.2	14.8	7.9	12.2	0.44
2010	153.3	143.0	61.3	43.4	37.0	19.7	15.3	21.9	9.9	30.1	0.40
July											
2005	205.8	45.3	145.1	31.9	101.8	22.4	36.5	8.0	6.5	1.4	0.70
2006	160.2	56.5	99.8	40.2	73.5	33.3	22.2	17.0	4.1	11.4	0.62
2007	182.0	51.8	142.0	51.3	95.5	40.8	40.3	27.6	6.2	12.2	0.78
2008	289.2	181.4	166.1	105.7	84.8	35.7	42.2	28.5	39.5	84.6	0.57
2009	193.7	160.5	122.0	80.0	78.7	41.4	25.0	17.4	18.3	45.7	0.63
2010	249.3	202.6	148.8	92.0	100.2	64.1	34.1	26.9	14.5	40.1	0.60
August											
2005	196.2	60.5	149.1	46.0	108.0	33.3	34.9	10.8	6.3	1.9	0.76
2006	226.6	102.6	101.7	58.8	69.7	47.8	25.6	18.1	6.2	11.4	0.45
2007	189.7	54.9	144.8	54.4	106.6	49.4	33.9	24.8	4.3	12.0	0.76
2008	270.8	199.5	194.3	147.8	117.0	69.6	44.9	42.5	34.0	72.5	0.72
2009	208.6	123.8	189.4	201.1	116.2	109.1	40.8	48.1	32.4	93.1	0.91
2010	346.9	185.8	263.6	152.4	136.3	46.1	60.4	49.4	66.8	164.2	0.76
September											
2005	202.3	70.0	40.5	14.0	27.9	9.7	10.5	3.6	2.0	0.7	0.20
2006	163.4	67.6	76.8	45.6	52.6	30.9	21.4	19.8	2.8	4.0	0.47
2007	164.4	61.4	32.9	27.2	23.1	21.9	8.9	7.9	0.9	1.1	0.20
2008	226.6	102.6	101.7	58.8	69.7	47.8	25.6	18.1	6.2	11.4	0.45

6.0 Elk Distribution and Burning at the Ya Ha Tinda

6.1 Background

Historical fire suppression, climate change-induced increases in fire frequency, and heightened human development in forest ecosystems have all led to escalated use of mechanical thinning in isolation or combined with prescribed fire to reduce the risk of fire (Schoennegel et al. 2009). The increase in fuels reductions and prescribed fires as fuel breaks in fire management systems has raised many questions about effects on wildlife. Unfortunately, not much is known about the effects of fuels reductions on wildlife (Pilliod et al. 2006). General relationships can be inferred from previous fire or forest harvest studies; for example, early seral species may benefit from decreased canopy cover following treatments (Pilliod et al. 2006). Fuels reduction projects are often implemented in different habitat and terrain conditions; wildfire and/or forest harvesting, and site-specific insights into the effects of these treatments are required to understand impacts on these various sites (e.g., Hebblewhite et al. 2009).

In 2006, Parks Canada began planned prescribed fires and mechanical fuels reductions adjacent to the Ya Ha Tinda grassland to achieve the dual objectives of: 1) increased facility protection of the Ya Ha Tinda ranch buildings and 2) facilitation of larger prescribed fire treatments in cooperation with Alberta Sustainable Resources Development to help increase habitat quality for focal species like elk, based in part, on previous research showing fire had positive effects on elk habitat quality and selection (Sachro et al. 2005, Hebblewhite et al. 2009). One of our objectives, therefore, was to monitor whether burning forests adjacent to grasslands at YHT altered elk use of the area.

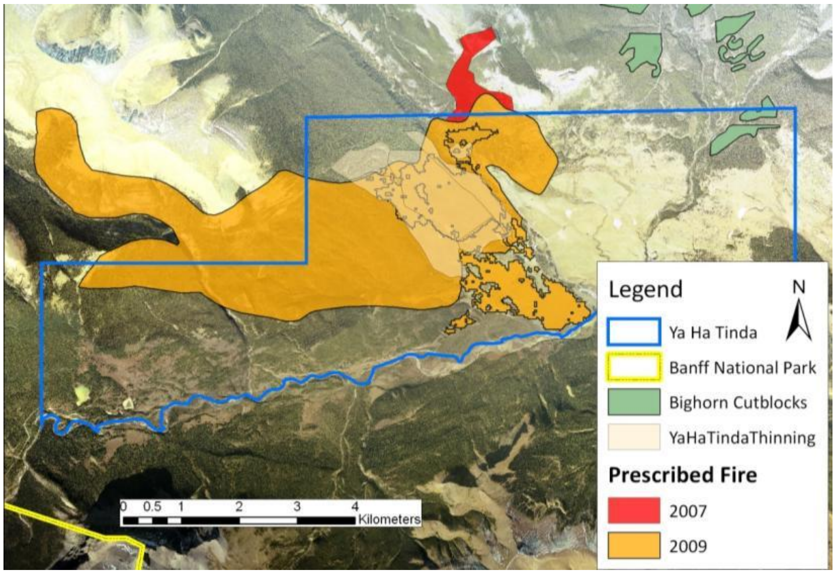


Fig. 6.1. Prescribed fire and fuel reduction thinning treatments at the Ya Ha Tinda ranch, Alberta, Canada, 2007 - 2009. Thinned areas were burned following fuel reduction in overlap areas. The Bighorn cutblock treatments (from the 1980s) are also shown for reference.

To examine distribution and responses in habitat use to these land use treatments (fuels reduction, burns, cut and burned) we used long-term pellet surveys. Future analyses will address habitat selection and distribution using GPS collared elk, but as of 31 March 2010, 7 of 10 elk GPS collars were still deployed in the field at YHT, and these data (these GPS collars are store on board) will not be available for analysis until winter 2010/2011.

6.2 Methods

In 2007, Parks Canada conducted mechanical thinning of mostly lodgepole pine and spruce forest west of Scalp Creek and just north of the ranch buildings on the southwestern slopes of Sheep Mountain, totaling a 373 ha area. They also lit a 139-ha prescribed fire in the same area north of the ranch buildings. These were followed up in the spring of 2009 by a 369-ha prescribed fire that burned a mix of lodgepole pine forests, grassland and shrublands west and north of Scalp Creek, and in the fall of 2009, 1,774 ha west of Scalp Creek. These 2009 burns covered approximately 340 ha of previous fuel reduction cutblocks. This created a mosaic of burned forest, burned grasslands, burned shrublands and burned-cutblock (cutblock) treatments to investigate elk responses during 2008 - 2011 (Fig. 6.1).

Pellet Counts

We continued long-term pellet counts in the grassland (<60% canopy cover; McInenly 2003) of the Ya Ha Tinda (Fig. 6.2a) as well as in transects established in spring of 2009 within forested and shrubby regions adjacent to the grasslands (Fig. 6.2b). Spring pellet counts were conducted during the first 2 weeks of May and represented winter use of the ranch. Fall counts occurred in the last 2 weeks of September and represent summer use. Grassland sites were 25-m^2 circular plots located in a systematic grid at 250-m intervals across the grasslands. Pellets in the forested areas were counted in transects measuring 2 x 100 m (200 m^2) (Fig. 6.2b). Of the 5 land cover types in 2011, burned forests represented 7.3% of all plots, burned grasslands 5.1%, burned cutblocks 18.2%, forests 25.2% and grasslands 44.3%.

Pellet groups were defined as containing at least 8 pellets and counted if >50% of the group was within the plot. Ungulate species recorded included elk, deer (*Odocoileus virginiana, O. hemonius*), horse (*Equis*), and moose (*Alces alces).* Color, weathering, and shape of pellets were used to determine pellet species and age. Elk pellets deposited in the winter had a squared bullet shape, while summer pellets transition to a soft coalesced or disc form (Murie and Elbroch 2005). Deer pellets were similar but smaller, typically under 1 cm in length. Black pellets were considered recently deposited, whereas grey or white color indicated pellets deposited last season or even a year earlier. The presence of wolf (*Canis lupus)* and coyote (*Canis latrans*) scat was recorded when encountered. To graphically represent a sample of years, we used an inverse-distance weighting interpolation ($1/d^2$) in ArcGIS with a fixed search radius of 300 m around the plots. After successful interpolation we set display breaks to six classes (Fig 6.6).

6.3 Results

Pellet counts indicated that less elk summered on the YHT grassland than wintered, particularly during the early 2000s (Table 6.1, Fig. 6.3). Decreases in number of winter elk pellet groups counted from 2000/2001 to 2008/2009 support aerial and ground counts that document a decline in the winter population size and an increase in the non-migratory summer population.

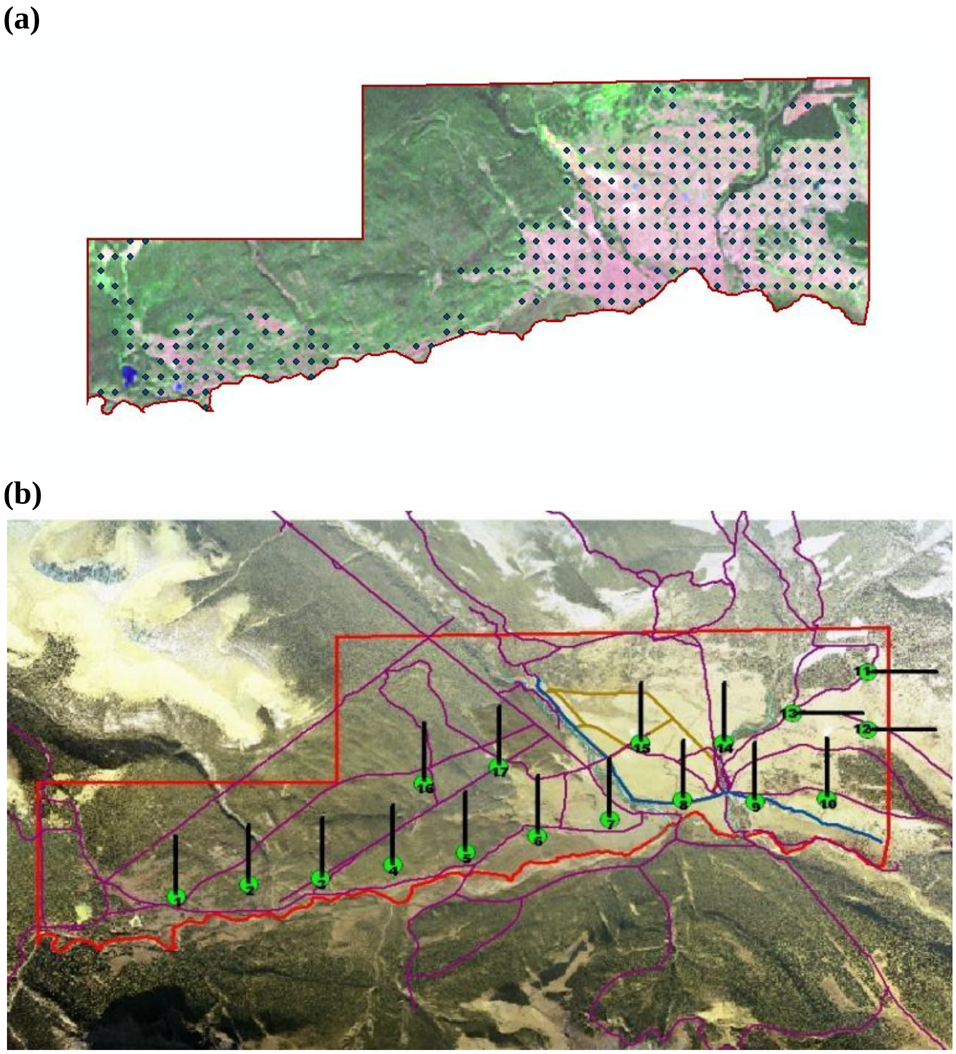
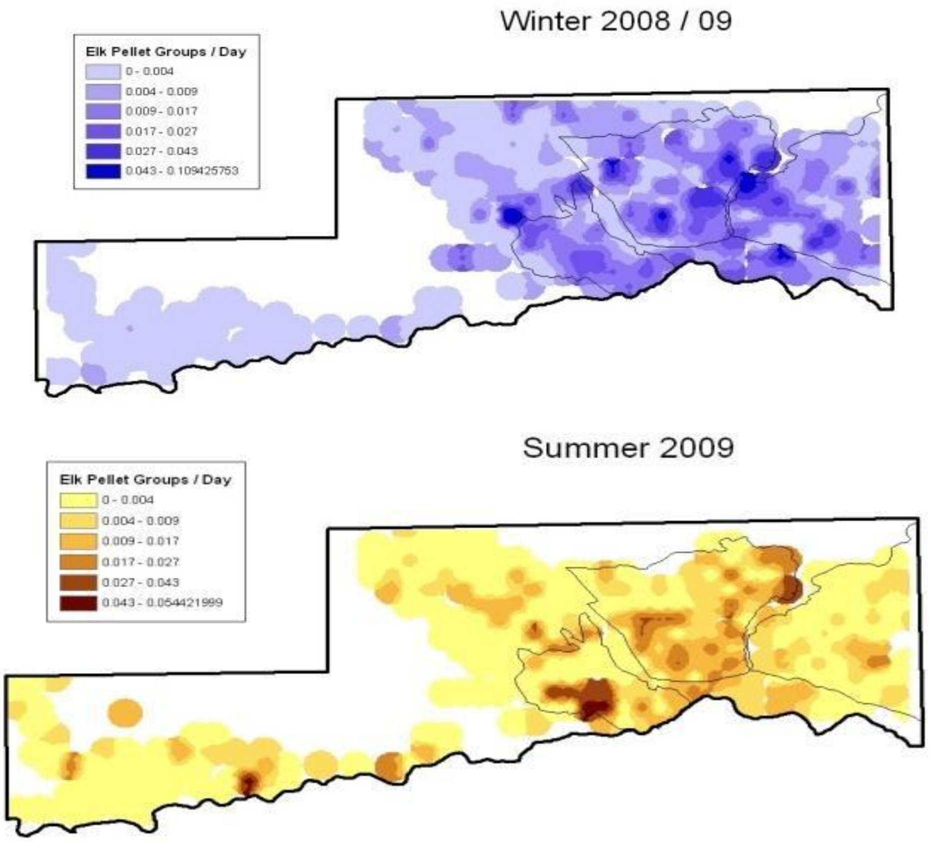


Fig. 6.2. Locations of (a) grassland pellet plots and (b) pellet transects within different habitats across the Ya Ha Tinda ranch, Alberta, Canada.

Table 6.1. Number of plots, minimum, maximum, mean, and standard deviation of past (McInenly 2003, Spaedtke 2009) and recent elk pellet groups counted, and deposition rates (No./day) observed during winter and summer elk pellet surveys at the Ya Ha Tinda ranch, Alberta, Canada.

Season	Year	N	Min	Max	Mean	S.D.	No./day	S.D
Winter	2000/01	270	0	24	3.01	3.33	0.014	0.020
Winter	2007/08	120	0	16	1.47	2.3	0.006	0.010
Winter	2008/09	195	0	25	2.25	3.03	0.010	0.013
Winter	2009/10	353	0	16	1.37	2.09	0.006	0.009
Winter	2010/11	356	0	19	1.15	2.10	0.005	0.010
Summer	2001	276	0	10	0.42	1.04	0.003	0.007
Summer	2008	266	0	9	1.10	1.74	0.008	0.010
Summer	2009	178	0	4	0.90	1.15	0.006	0.008
Summer	2010	379	0	18	1.39	2.29	0.010	0.007
Summer	2011	351	0	6	0.43	0.89	0.003	0.007

Fig. 6.3. Exmapl of changes in mean elk and deer pellet counts in winter and summer Ya Ha Tinda ranch, Alberta, Canada.



TO BE FINISHED

7.0 Aversive Conditioning of Elk

7.1 Background

One consequence of declining migration is an increased elk population residing on the Ya Ha Tinda winter range during the summer months, instead of migrating into Banff National Park (Hebblewhite et al. 2006). Resident elk populations were increasing to the point where potential impacts on rangeland productivity were being detected by Parks Canada, and so investigations into management strategies to reduce elk residence on the Ya Ha Tinda winter range began in 2004. Following from experiences with aversive conditioning in the Banff town site (Kloppers et al. 2005), Parks Canada initiated an Aversive Conditioning (AC) program on the YHT grassland conducted as a thesis (Spaedtke 2009). The goals of the AC program were to: 1) re-establish migratory behavior of the resident elk and 2) reduce grazing pressure by elk to increase biodiversity in the grassland ecosystem (Spaedtke 2009). Spaedtke (2009) used controlled movement of 2 - 15 people on horseback to condition the elk off grazing land into habitat adjacent to the main YHT winter range pasture. Spaedtke (2009) showed that while AC was successful at changing the summer distribution of elk, and therefore reducing grazing pressure, there was no increase in long-distance migration associated with AC. The AC program was continued intensively by Spaedtke through 2007 and then at a reduced intensity by Parks Canada from 2008 – 2010, and then discontinued in summer 2011.

One of the questions emerging from this shift to less intense AC was whether the response of elk was any different than that of the earlier (2005 - 2007) period, and what the response will be following the cessation of AC in 2011. Therefore, we conducted two analyses to address these questions, notwithstanding that we are still collecting data on elk distribution during summer 2011. First, we tested for changes in the cumulative home range of all resident GPS collared elk at Ya Ha Tinda between 2004 (no AC) and 2005 (first year of AC) to enable the future comparison of 2005 to 2010 (AC) to 2011. Second, we tested for temporal responses in the distribution of elk on the Ya Ha Tinda grassland following the zonal approach of Spaedtke (2009) whereby proportional use of different grassland zones were compared within and between years. To determine the herd’s response to AC, the data were separated into pre AC (2002-2004), during AC (2005-2010) and post AC (2011). By comparing elk distribution pre AC, 2004, and during AC, 2005, it is possible to quantify how effective AC was at changing the resident population’s home range.

7.2 Methods

Home Range Comparison

We defined the cumulative home range of all GPS collared resident elk using a fixed-kernel density (Seaman 1996) with a reference bandwidth estimator to define the YHT herd’s 2004 and 2005 summer home ranges. To ensure all migratory elk had moved off the winter range and would therefore not alter the expanse of the resident home range, we defined summer as 1 June

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through 31 August. A constant 973 points (the minimum number of points for a GPS collared elk) were randomly selected from each GPS-collared resident for both 2004 (no AC) and 2005 (first year of AC). All analyses were conducted in ArcMap (ESRI, Redlands, CA). A single GIS raster was calculated for the sum of the Utilization Distribution from the kernel density estimator for all resident elk home ranges for each year. A single measure of overlap between the two time periods as a full function of both Utilization Distributions (UD) was determined using Bhattacharyya's affinity (BA), which is calculated as the product of the two Utilization Distributions (Fieberg and Kochanny 2005).

Spatial Distribution in Response to Aversive Conditioning

We used all GPS and VHF data collected at the YHT from 2002 - 2010 (2011 will be added in the future) to test for shifts in spatial distribution across time and within years, and tested for differences in the spatial distribution within Spaedtke's (2009) concentric 1-km circles centered on the main ranch pasture at YHT (Fig. 7.1). Zones are labeled zone 1 (center) to Zone 6, which includes everything beyond the outer circle. We then compiled both GPS and VHF radio-telemetry data for each season (May - Aug), 2002 through 2010. The GPS data were standardized to two hour intervals. No GPS data were available for summer 2008, and the GPS data were not complete for summer 2009 and 2010 (some store on board collars are still deployed), so those years are represented only with VHF data. All the data were projected onto the circles in GIS and selected by month, year, and zone. We then calculated both percent of use within a zone by months and across months, for the entire population with respect to year. These percentages were averaged within the year to yield a percent of zone use for each year.

Because the methods for the AC program involved pushing the elk off target grasslands starting in the morning and usually finishing around midday, we also compared resident elk distribution separately for day and night. To determine zone use with respect to when the grazing area is most used, we used the concentric circle layout (Spaedtke 2009). Using 2-hour standardized GPS data from the summers (May - Aug) of 2002 through 2007 and summer 2009, we separated the data into day (0600-1800 h) and night (1800-0600 h). We then plotted the GPS data for residents onto the circles and selected the points by year, time, and zone. To compare across years, we calculated percent by day and night of points in each zone with respect to the total number of points in that year. To understand how zone use changed with regard to day and night during AC, we separated the years into Preconditioning (2002 - 2004) and During Conditioning (2005 - 2007, 2009). Then we separately averaged zone use during the two periods (Fig. 7.3).

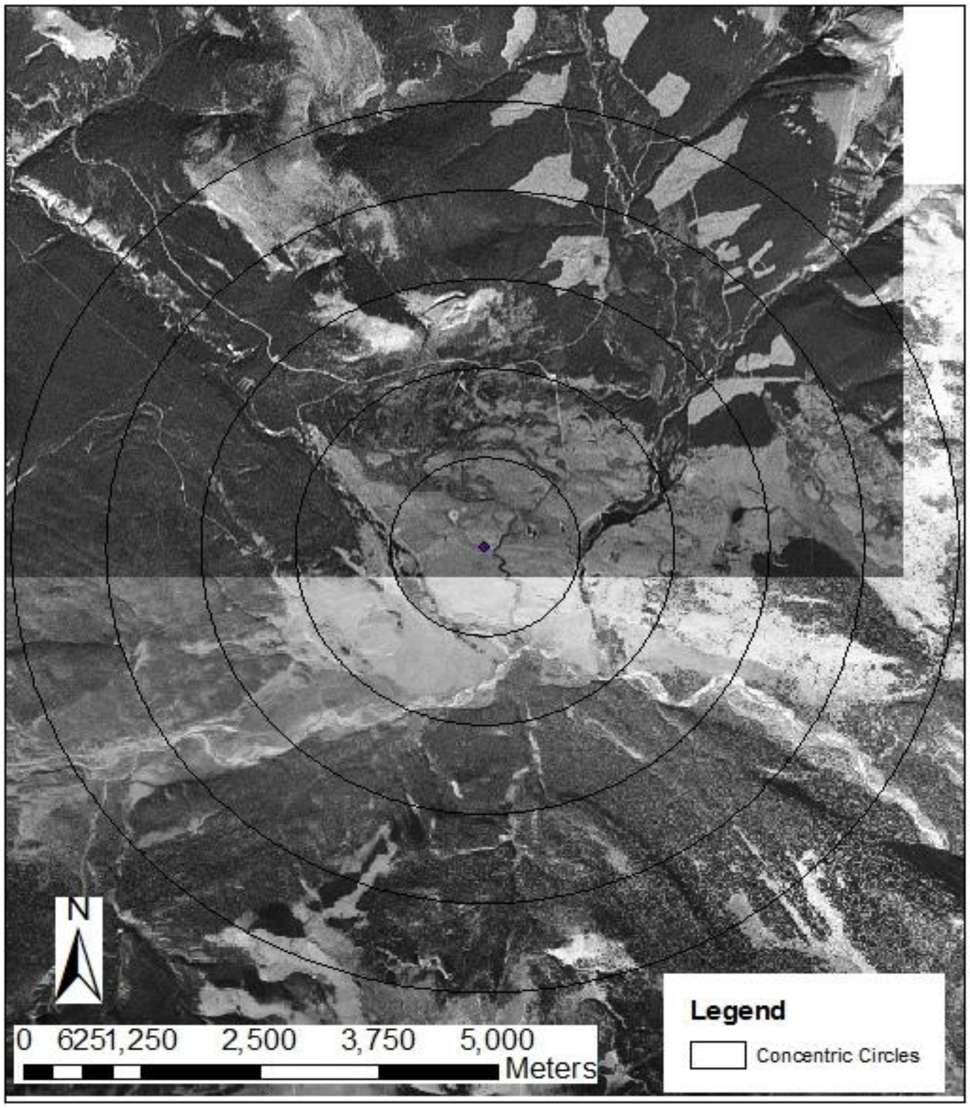


Fig. 7.1. Concentric Circles centered on the target YHT grazing land, Alberta, Canada, with an increasing radius of 1 km.

7.3 Results

Home Range Overlap

Following methods laid out by Fieberg and Kochanny (2005), we calculated three-dimensional overlap of two UDs using ArcMap (Fig 7.2). The resulting overlap between the home ranges for 2004 and 2005 was 37.8%. Referencing the 95% minimum convex polygon (MCP), the change in home range had a distinct direction, indicating that AC was successful in re-establishing westward migratory behavior, by at least a few elk. This information will be useful for future assessments regarding the change in home range between the last year of AC, 2010, and the first summer post AC, 2011. For example, if elk revert back to their pre-AC distribution in 2011, we might expect overlap between 2005 - 2010 and 2011 to be close to the 37% we observed here.

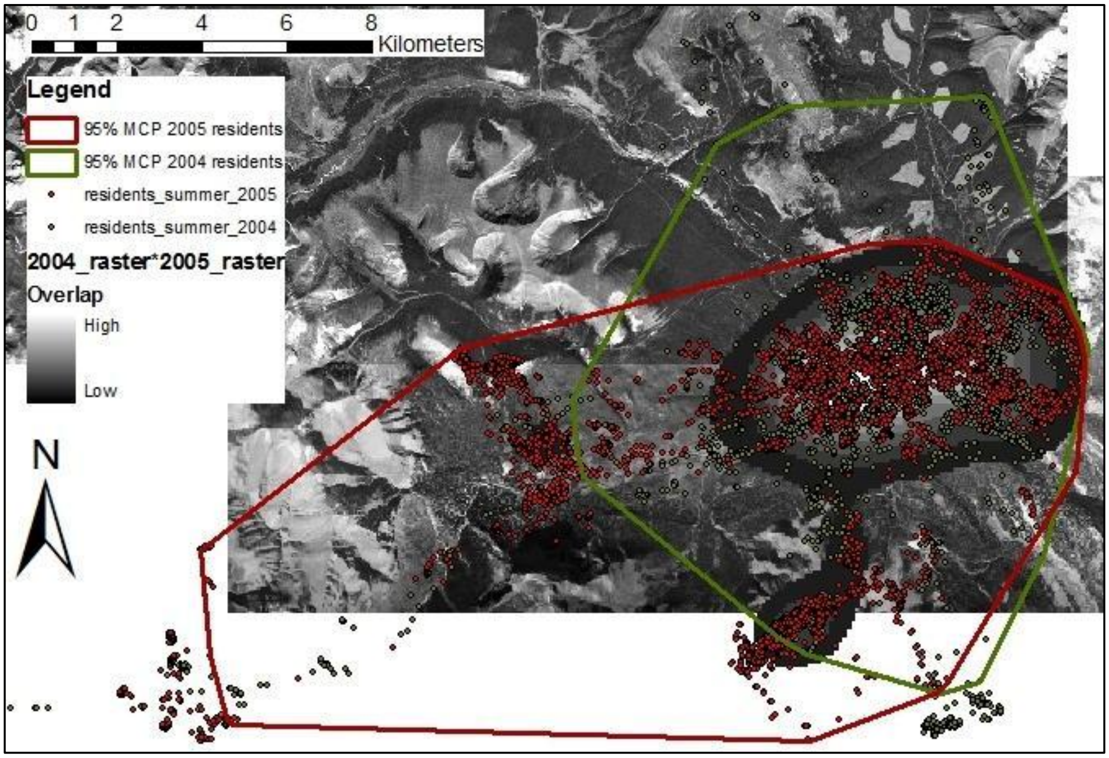


Fig. 7. 2. Locations of 2004 and 2005 residents on YHT grasslands, Alberta, Canada. The resulting overlap between two summers was calculated at 37.83%.

Spatial Distribution in Response to Aversive Conditioning

Results from comparing percent use across years yielded data trends different than expected (Fig. 7.2). We found an average increase in use of zone 1 by 5.3% between 2003 and 2010. The data show a decrease in collared elk use of zone 6, and more use of all inner circles. Our results have no real trends that suggest Aversive Conditioning had any significant effect on the way the entire YHT elk herd use the grasslands. Some caution is warranted, however, because we have not yet analyzed the unrecovered GPS data stored onboard collars that cover the period from 2009-2010.

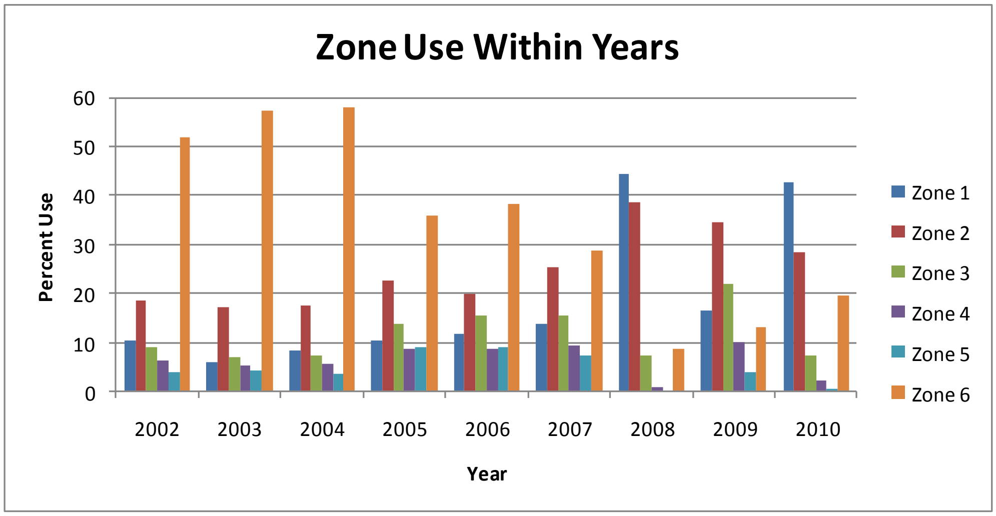


Fig. 7.3. Percent zone use by Ya Ha Tinda elk, Alberta, Canada, calculated with respect to the entire population (VHF and GPS data) within the summer season each year. 2008 and 2010 are represented with VHF data only.

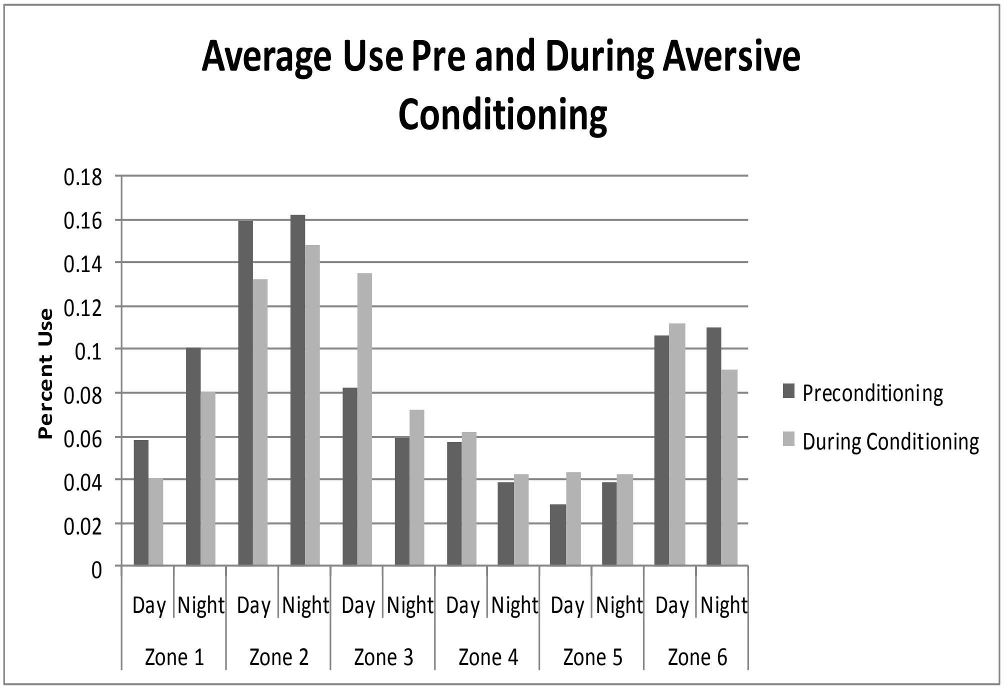


Fig. 7.4. GPS-collared resident elk use of grassland zones, Alberta, Canada, during day (0600 – 1800) and night (1800 – 0600), averaged across Preconditioning (2002 - 2004) and During Conditioning (2005 - 2007, 2009).

Comparing zone use by resident elk between day and night showed a greater use of zones closer to the target grassland during night (Fig. 7.4). During Preconditioning years, zones 1 and 2 were used on average 2.31% more during the night than the day. During Conditioning years showed averaged 2.76% increased use of zones 1 and 2 during night. The trend reverses for zones 3 and 4, which showed greater day use in the During Conditioning period. This suggests the elk were pushed off zones 1 and 2 during the day into zones 3 and 4. After being pushed off zones 1 and 2 during the day, elk headed back to the inner zones at night. Nonetheless, the results do show a decrease in total use for zones 1 and 2 during conditioning indicating that AC did reduce resident time on the inner zones. AC did not appear to have any substantial effect on resident use of zone 6.

7.4 Discussion

Similar to Spaedtke (2009), we showed a strong shift away from the YHT grasslands between 2004 and 2005, that echoed his conclusions that AC may have had some success at increasing long-distance movements off the YHT, if not necessarily migration per se. Our home range analysis approach will allow us to test for differences in elk distribution after the cessation of AC in 2011. However, contrary to Spaedtke (2009), who reported a strong seasonal shift of elk away from grasslands during his study (2005 - 2007), we found weaker evidence across the entire AC time period (2005 - 2010) for a strong shift in spatial distribution away from targeted grasslands at the center of the YHT pastures.

The unexpected results could be related to several factors. There was substantial variation in sample size between years, and thus percentages could be skewed and not accurately represent the population. Another problem is the challenge of classifying migrants and residents consistently over the duration of the project. We may have inadvertently classified some migrants as residents and included them in the early and late months of a year in the zonal analysis, whereas Spaedtke (2009) only used residents for the entire season. Future analyses, which are underway (Bunnefeld et al. 2011), would benefit from consistent definitions of migrant and resident elk. Finally, a bias may have been introduced with the inclusion of VHF data for years with no GPS data (2008) or for which GPS data have not been recovered yet (2010). VHF data is collected on foot and based mainly on the ranch, with a large proportion of that sample collected during morning observations along the ranch road. The result is a biased percentage of VHF locations, possibly showing more herd use of inner zones than is accurate. The results as a whole have shown AC did have an impact on resident elk, but may not have reduced the percentage of the herd's use on YHT grasslands over the entire duration of the AC period (2005-2010). These results will be an important reference in future analyses comparing the YHT elk herd's response to the end of AC.

8.0 Long-term Elk Population Dynamics at the Ya Ha Tinda

8.1 Background

Long-term population studies of individually-marked ungulates are critical to support population management and scientific advancement (Gaillard et al. 2000). Alberta has had a long history of supporting long-term studies on bighorn sheep (*Ovis canadensis)* in the Sheep River, Kananaskis country (Festa-Bianchet 1988, Ross et al. 1997, Ruckstuhl et al. 2003, Coulson et al. 2005) and Ram Mountain (Jorgenson et al. 1997, Coltman et al. 2003, Festa-Bianchet et al. 2006*b*, Pettorelli et al. 2007) and mountain goats (*Oreamnos americanus*) in the Caw Ridge area (Cote et al. 1998, Cote and Festa-Bianchet 2001*a*,*b*, Hamel et al. 2006). Combined, these studies have directly contributed to management for these two key ungulate species both in Alberta and across North America. For example, information provided by these studies has provided critical direction to trophy management, especially of bighorn sheep (Jorgenson et al. 1998*a*, Coltman et al. 2003, Coltman et al. 2005).

Since 2001, in collaboration with project partners C. Cassidy-St. Clair (University of Alberta) and C. White (Parks Canada), we have monitored 226 radio-collared adult female elk to determine how changes in adult elk population dynamics are affected by humans (e.g., harvest, habitat management), natural factors (e.g., predation, climate), and vegetation management. Alberta Fish and Wildlife have also collected long-term population data since 1972 on population size and calf recruitment. Combined, our studies of elk population dynamics at Ya Ha Tinda represent one of the longest-term population studies of elk in a system with intact natural predators including wolves (*Canis lupus*), grizzly bears (*Ursus arctos*), and human hunting (Hebblewhite et al. 2005, Hebblewhite et al. 2008, Hebblewhite and Merrill 2008). Our long-term objective is to continue the past population monitoring while assessing short-term management actions on elk responses. Additional monitoring of wolf packs in this area is planned by Parks Canada.

Our long-term research objectives focus on the following key questions:

1) How does adult female survival and hence population dynamics change as a function of climate, predation by wolves, and human harvest regimes?

2) What are the most sensitive population parameters (adult female survival, calf survival, pregnancy, etc.) in elk populations under heavy predation?

3) How do long-term changes in habitat through restoration of fire contribute to changes in elk populations (see section 7 above)?

4) How do changes in migration influence long-term population dynamics of the Ya Ha Tinda elk herd?

5) How can we focus on addressing the effects of differing harvest regimes on population genetics of elk, migrant vs. resident elk population genetics, and long-term changes in population structure?

In this final report, we focus on summarizing results of objectives 1 – 4 above. Longer-term studies to address objective 5.

8.2 Methods

Elk Capture and Handling

Elk were captured using a corral trap, aerial net gunning, or ground darting. Corral trapping was the most common capture method. We employed two different large corral traps during 2001 - 2011 in a total of three different trap site locations. The two traps were the 'red' trap on loan from Rocky Mountain Elk Foundation-Canada, and a new "green" trap on loan from RMEF/AB-SRD through Elk Island National Park (Table 8.1).

Table 8.1. Locations, IDs, and active years of elk corral traps at the Ya Ha Tinda, Alberta, Canada, 2001 - 2011.

Trap Site	Trap ID	Years	UTM Easting	UTM Northing
East side of Scalp Crk	Green Trap	2001/02 – 2003/04	601050	5733450
Main Pasture	Red trap	2001/02 – 2007/08	599650	5733350
West of Mares Ridge	Red trap	2008/09 – 2010/11	597850	5732650

Traps were usually baited with bales of second-cut alfalfa hay. During each trapping period the trap was pre-baited with 3-4 square bales inside the trap and 2-3 bales spread outside the trap to lure the elk inside. During 2008 - 2011, after the elk trap was moved to the original Morgantini trap site west of Mares Ridge, ranch and research staff would occasionally herd elk on horseback to the general vicinity of the trap to facilitate capture. The trap doors were closed using a manual counter-weighted trigger mechanism. A portable ground livestock scale (0 -5,000 lbs) was set inside the last chute closest to the squeeze tub with a plywood platform on top to weigh elk. Elk were moved from the corral to the holding pens, onto the scale, and then into the squeeze where they were blindfolded, haltered and restrained.

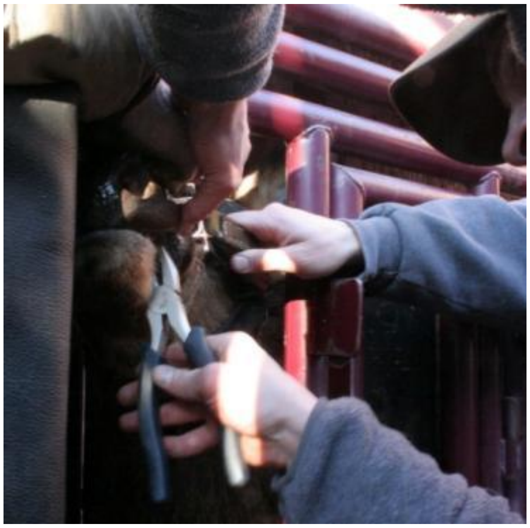
Elk (bl288) with ear tags in the corral trap

Once restrained, elk were processed following animal care and use protocols reviewed and approved by the University of Alberta (IACUC # 611/11/11), University of Montana (University of Montana IACUC protocol: 059-08MHECS-120908), and Parks Canada Animal Use Committee (Permit #YHTR-2009-2875). Restrained elk were then ear-tagged (All-flex^{TM}, Super

Maxi size) for identification, examined for injuries, and radio-collared. Radio collars were fitted to adult and subadult (2+ years) females with VHF (LOTEK LMRT-4, expected collar life of 4 - 6 years, 800 g) or GPS collar (LOTEK models 2200, 3300, 4400, expected battery life 1 - 2 years depending on collar programming, 1500 g). Since 2002, we adopted a 1 - 3 color banded-tape coding scheme to make radio collars more readily identifiable from a distance. Body weight (kg) was recorded with the portable scale, and we recorded body condition using a previously developed index (Cook et al. 2004) and physiological measurements (temperature/pulse if overheating). Whole blood and hair samples were collected for serum pregnancy and pathogen testing, and DNA sampling.

Estimation of age-specific survival rates for known-age individuals is a fundamental component of understanding population demography in any long-term population study. Numerous long-term studies of ungulate population demography have estimated age by extracting an incisiform tooth and using cementum annuli rings (Nelson 2001). Regardless of concerns over tooth extraction (Festa-Bianchet et al. 2002), in elk the vestigial canine has no foraging purpose, and is easily removed using dental extractors with no expected negative effects. Starting in 2008, we started collecting the vestigial canine for aging from captured elk following approved amendments (University of Montana IACUC 059-08MHCES-120908, IACUC # 611/11/11) to our animal care protocols adopted from Mansfield et al. (2006) in consultation with Dr. Todd Shury, DVM (T. Shury, pers. comm.).

Elk were occasionally captured using aerial net gunning or ground darting to augment capture efforts when ground captures were unsuccessful or specific elk needed to be captured. Capture by aerial net gunning was conducted by Bighorn Helicopters and all procedures were consistent with the Alberta Wildlife Animal Care Committee Class Protocol #008 (Alberta Fish and Wildlife Division 2005). Baseline capture mortality rates for helicopter net gunning conducted by Bighorn Helicopters from 2001 - 2005 during this study were <4%. Aerial chases of elk were ≤5 minutes. Once successfully netted, Bighorn Helicopters capture crews quickly subdued the elk, and followed handling protocols described above. On the rare occasions we ground-darted elk, we used a CO_2 powered Pneu-DartTM remote drug delivery dart gun system (see Cattet et al. 2005) using the lowest possible charge (e.g., brown charge, Jessup 2001) to further reduce impact injuries. Elk were immobilized using either: 1) a combination of Carfentanil (WildnilTM) and Xylazine (RompunTM) at a dosage of 0.01 mg/kg and 0.2 mg/kg respectively (reversed by Naltrexone at a dosage of 125 mg Naltrexone/mg of Carfentanil administered) or 2) a combination of Telazol:Xylazine (Millspaugh et al. 1995, Walter et al. 2005) at a dosage of 3.0 mg/kg and 1.5 mg/kg, respectively (reversed using tolazoline at a dosage of 2.0 mg/kg body weight). If required, we used 2 mg/kg IM of Ketamine to extend immobilization of elk.



Canine extraction in a restrained elk

Elk Radio Telemetry and Mortality Monitoring

Radio-telemetry monitoring of radio-collared elk for spatial distribution and mortality monitoring occurred in two distinct study periods with slightly different protocols and intensities during summer (1 May - 31 Oct). During the first study period (Hebblewhite 2006) from 2001-2005, elk were relocated during summer once every 6 to 9 days for a total of ≥10 locations per summer per individual (Hebblewhite et al.2006). Between the period from 2005 - 2007 (Spaedtke 2009) and 2008 - 2010 (this study) intensive summer monitoring of migrant elk was reduced to focus research on aversive conditioning of residents. As a result, from 2006 - 2009, summer monitoring efforts were focused mainly on the resident portion and only two annual summer telemetry flights were conducted on ~ 1 July and again during mid-October of most falls. Winter monitoring has remained unchanged since the beginning of the study in 2001/2002, where efforts were assumed to be equal for migrants and residents during the winter months, and each elk was relocated using telemetry at least on a weekly basis from the ground, and almost daily for mortality monitoring. In the summer of 2010, monitoring for migrant elk was reinitiated using the same protocol and similar schedule as the 2002 - 2004 study. GPS collars were programmed to obtain locations anywhere from every 15 minutes to 4 hours throughout the study, and lasted anywhere from 2 months (collar failure) to 2 years. See Hebblewhite et al. (2007) for details of GPS collar performance during the 2001 - 2005 period of the study. Mean VHF location error was 218 m, and using the Bessel function to model GPS error, revealed 50% of the locations were within 34 m of the true location, and 95% within 113 m (Hebblewhite 2006).

Adult Female Elk Mortality

Mortalities have been detected using ground and aerial telemetry, and mortality signals investigated on the ground or via helicopter as quickly as possible. Cause of death has been ascribed to predator (e.g., wolf, grizzly, cougar), human-caused (e.g., hunting, accidental), disease, starvation, and unknown causes using the same methods of Hebblewhite and Merrill (2011). Deaths within 1 month of capture (n = 2 over the last 10 years total) were considered capture myopathy and censored.

Cow:calf Recruitment and Natality Rates

For all observations of groups of collared, tagged, and/or un-collared elk, we recorded time, date, location, and the numbers of tagged elk in the herd, whenever possible. We used standardized criteria (Smith and MacDonald 2002) to sex- and age-classify elk in groups to obtain demographic data such as cow:calf ratios, and other behavioral observations. Observations were made from sufficient distances to eliminate the risk of disturbing elk. Here, we examined trends in recruitment from 2001 - 2011 by defining recruitment as the calf ratio in late winter (Feb – Apr), and natality as cow:calf ratio in summer (June – Aug) following Hebblewhite (2006, Appendix 1B).

We estimated the cow:calf ratio for both periods following Czaplewski et al. (1983) according to

$$Y_{ij} = \frac{\sum_{i=1}^{n} calves_i}{\sum_{i=1} cows_i} \qquad \text{Eq. 1}$$

where i = 1 to n elk herds classified within season-year j, i.e., 2011 recruitment. We calculated the standard error in Y_{ij} assuming errors were binomially distributed following Czaplewski et al. (1983) according to:

$$SE = \sqrt{\frac{Y_{ij}(1-Y_{ij})}{k_{ij}}} \qquad \text{Eq. 2.}$$

where Y_{ij} is the cow:calf ratio for season-year j, and $k_{\mathrm{ij}} = \sum_{i=1}^{n} calves_i + \sum_{i=1}^{n} cows_i$, namely, the total number of elk counted in any given season-year (Czaplewski et al. 1983). Furthermore, we compared observed recruitment and natality rates from 2001 - 2011 to earlier rates estimated by Hebblewhite (2006) to understand long-term trends in recruitment and natality in the Ya Ha Tinda elk herd.

Winter and Summer Aerial Elk Surveys

Parks Canada and/or the Alberta Fish and Wildlife Division have conducted aerial surveys in rotary-wing aircraft (Bell 206B Jet Ranger) every winter since 1972 except in 1981, 1986, and 1992, and approximately every third summer since 1977 (1977, 1978, 1979, 1980, 1982, 1983, 1984, 1985, 1987, 1991, 1998, 2003, 2004). Here, we only report results from winter surveys. See Hebblewhite et al. (2006) and Spaedtke (2009) for details of more recent summer aerial surveys. Winter aerial surveys were conducted 1 – 200 m above ground level at 50 – 70 km/hr. Winter aerial surveys were flown 1 – 2 days after heavy snowfalls during morning (0800 – 1200 h) on sunny or flat light days during January or February to maximize sightability of elk (Allen et al. 2008). Large herds (>50) were photographed for counting. Continuous participation from 1972 – 2001 by one ABF&W biologist ensured data consistency. After 1977, herd size, general herd composition (bull, female, mixed), activity, and location were recorded and later transcribed to UTM coordinates. Locations were considered accurate only to 500 m because of mapping differences over time. Agency biologists occasionally conducted surveys in another agency's jurisdiction. When surveys overlapped in the same year, we used only agency area-specific data.

Migratory Behavior

We examined trends in migratory behavior by calculating the population-level migrant:resident (M:R) ratio using the maximum number of elk observed from air and/or ground during summer on the YHT winter range as a proportion of the following winter's aerial count. A more quantitative analysis of classifying animals is under analysis (S. Eggeman: MSc Thesis, University of Montana).

Adult Female Elk Survival Analysis

To estimate adult female survival, we organized the long-term VHF telemetry and GPS databases representing 226 individuals and 113 deaths into continuous long-format where each row represents a location for a marked individual and the columns represent relevant factors affecting survival (Cleves et al. 2010). We defined mortality events (mortality = 1, alive = 0), 2 seasons winter = 0 (1 Dec – 31 May), summer = 1(1 Jun - 31 Nov), and migratory status (migrant = 1, resident = 0) (Hebblewhite and Merrill 2011). We used a study-based staggered entry design (Pollock et al. 1989) setting origin of analysis time from the date of first capture, 2 February 2002 (Fieberg and DelGuidice 2009). To standardize location intervals for GPS data, collected in 15-minute to 2-hour intervals, with VHF data, we re-sampled GPS data at the same average interval time as VHF locations (Hebblewhite and Merrill 2011).

We tested for differences in survival between migrants and residents using two complimentary methods. First, we used a non-parametric Kaplan-Meier (K-M) approach to estimate survival (Hosmer et al. 2008, Cleves et al. 2010). We calculated estimates of total survival for migrants and residents as well as mean annual survival using STATA 10 (StataCorp LP, College Station, TX). Annual survival estimates were calculated from 1 June to 31 May of the following year. Seasonal K-M estimates will be calculated from 1 May to 31 October for summer and 1 November to 30 April of the following year to represent winter. We chose these seasonal dates to capture summer migration events and winter mortality events separately to compare seasonal survival rates (Hebblewhite et al. 2006). We will compare K-M survival estimates for migrant and resident elk using the log-rank and Wilcoxon tests (Hosmer et al. 2008, Cleves et al. 2010).

To address long-term objective 3, which demographic traits were driving population growth rate of elk , we constructed pre-birth pulse age-structured elk matrix models using demographic data from 2001 - 2005 (see Hebblewhite and Merrill 2011 for details). We modeled elk life history in 5 age classes, closely following Raithel et al. (2006): calf (<1 year old), yearling (1 - 2 yrs), prime-age adult (2 – 10 yrs), old-age adult (10-15 yrs), and senescent (>15 yrs). We divided adults into prime-age and old-age to account for survival declines prior to senescence (Flook 1970, Gaillard et al. 2000). Reproduction occurs at ages $\geq$2 years. Calf survival was assumed to be constant between sexes, and sex ratio at birth was assumed 50:50. Fecundity is the product of the number of female calves born, intrauterine survival, and pregnancy rate. Elk were assumed to bear one young, and because pregnancy was estimated on 8 March, late-term intrauterine survival was also assumed to be 1 (Raithel et al. 2007). Thus, fecundity was simply half the pregnancy rate. We used vital rates estimated from 2001 - 2005 following Hebblewhite and Merrill (2011).

Life-stage simulation analysis (LSA) was used to explore the effects of vital rate variation in population growth rate and elasticity (Wisdom et al. 2000). LSA simulates replicate matrix models using vital rates drawn at random from a mean and process variance (Wisdom et al. 2000). Process variance is estimated using variance components decomposition from time-series or across point estimates (White 2000). We used mean vital rates for elk in our study as defined above. However, given the short-time span of this study (3 years), we did not estimate process variance for our study, but used the *within-study* process variance estimates ($\hat{\sigma}^2_{within}$) from Raithel's (2005) comprehensive review. Given these vital rates and process variance estimates, 1000 replicate matrices were simulated from the observed distribution for each vital rate using the limitsens.m MATLAB code from Morris and Doak (2002) modified for elk by Raithel et al. (2006).

Pregnancy Rate

Adult female pregnancy rates were estimated from blood samples collected during winter trapping by measurement of pregnancy-specific protein-B (PSPB; BioTracking LLC, Sasser 1998).

8.3 Results

Elk Capture and Handling

A total of 491 elk were captured between 2001 and 2011 at the YHT as part of long-term monitoring. On average, we captured 48.8 elk/year (Fig. 8.1, Table 8.2), ranging from 9 – 121/year. Not all captured elk were radio-collared, and there were also 96 re-captured elk over the course of the study. We radio-collared 226 unique individual elk a total of 324 times from 2001 - 2011 (Table 8.2). Re-collared animals were most often GPS-collared elk whose collar battery life had expired and were subsequently replaced with a VHF collar, but also VHF-collared animals that were known residents that later fitted with a GPS collar for Spaedtke's (2009) research. Most elk (81%) were captured by corral trapping, with 17% captured via helicopter net gunning, and only 2% (5 animals) captured by ground darting (Table 8.3).

Table 8.2. Summary statistics for elk captured and radio-collared during winter at the Ya Ha Tinda ranch, Alberta, Canada 2001-2011.

	Captured	Collared	Re-captured	Re-collared	Unique Individuals
Total[1]	491	324	96	73	226
mean	48.8	32.1	9.6	7.3	22.5
SD	37.20	17.61	7.47	6.46	14.11
min	9	9	0	0	0
max	121	57	20	19	44

1- Does not include 3 radio-collared immigrant elk

Table 8.3. Summary of capture methods from 2001 - 2011 for elk captured during winter at the Ya Ha Tinda ranch, Alberta Canada.

Year	Corral Trap	Ground Dart	Heli-Net
2001	37	0	5
2002	50	1	6
2003	49	0	4
2004	17	0	5
2005	15	4	9
2006	27	0	15
2007	9	0	0
2008	38	0	3
2009	0	0	9
2010	16	0	0
Grand Total	261	5	56
Percent	81	2	17

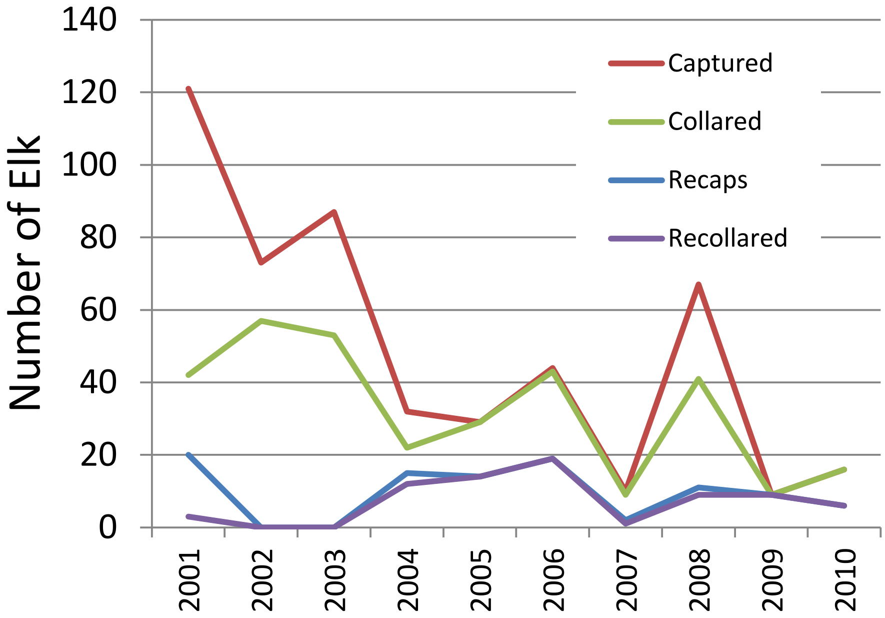


Fig. 8.1. Summary of elk captured, radio-collared, and re-captured from 2001 - 2011 during winter at the Ya Ha Tinda ranch, Alberta, Canada.

Elk Radio Telemetry

We monitored a total of 226 unique collared adult female elk from 2002 - 2011 in the YHT herd. On average, we had 90 adult female elk radio-collared per year, with 81 VHF collars/year and 9 GPS collars/year, with a range of 4 - 25 GPS collars deployed in any one year (Table 8.4). Because some elk wore both GPS and VHF collars at different times during their monitoring, the total numbers of unique VHF and GPS-collared elk are not independent (Table 8.4). On average, individual elk were collared for an average duration of 3.1 years. From VHF-collared elk, we obtained an average of 18 (range 9 -51) VHF locations/elk/year, with an average of 8.5 locations during winter and 9.6 locations during summer (Table 8.4). For the GPS-collared elk, we deployed GPS collars once on a total of 56 unique elk, collecting an average of 4,848 locations/elk, and a total of 431,664 GPS locations in total. On average, fix success rates were ~92%, similar to those reported by Hebblewhite et al. (2007) of 90.1%. Also, the distribution of VHF and GPS-collared elk closely matched each other on both the winter and summer ranges (Fig. 8.5), and the total area covered by YHT elk encompasses 5,472 km^2 and did not changed from the earlier 2002 - 2005 period (the elk study area polygon in Figure 8.5 is the same as described by Hebblewhite et al. (2006).

Table 8.4 Summary radio-telemetry table for VHF and GPS-collared elk from 2001 to 31 March 2011 in the Ya Ha Tinda elk herd, Alberta, Canada. The table shows total number of elk collared/year, number and average number of VHF/GPS locations/individual elk, and total number of locations. Note that the total number of VHF and GPS-collared elk do not add up because some elk wore both kinds of collars.

Year	# Elk Collared	Total VHF Locations	Total # VHF-collared	Mean VHF Locations/ Elk	Total # GPS-collared	Total GPS Locations	Mean GPS Locations/ Elk
2002	41	1903	37	51	4	11,192	2798
2003	81	2514	73	34	8	36,342	4543
2004	99	1875	74	25	25	88,152	3526
2005	92	963	81	12	11	51,498	4682
2006	113	1375	99	14	14	126,342	9024
2007	103	861	94	9	9	86,926	9658
2008	81	1023	81	13	0	0	
2009	108	1259	101	12	7	27,157	3880
2010	97	930	91	10	6	4,054[2]	676
2011	87	357	81	4[3]	6	0[2]	
Average	90.2	1306.0	81.2	18.6	9.0	43166.3	4848.3
Totals	226[1]	13060	192[1]		56[1]	431663	

[1] Represents total # of unique individual elk radio-collared. [2] 2010/2011 GPS collar data is still being collected from store-on-board GPS collars on 6 elk. 2011 data only up to 31 March 2011.

Table 8.5. Seasonal distribution of VHF and GPS collar data between summer (May – Oct) and winter (Nov – Apr) on adult female elk in the Ya Ha Tinda elk herd, Alberta, Canada, 2002 - 2011.

	VHF			GPS		
Year	Summer	Winter	% Winter	Summer	Winter	% Winter
2002	966	937	49.2%	8278	2914	26.0%
2003	1381	1133	45.1%	21891	14451	39.8%
2004	1043	832	44.4%	63400	24752	28.1%
2005	663	300	31.2%	45134	6364	12.4%
2006	573	802	58.3%	92759	33583	26.6%
2007	515	346	40.2%	69521	17405	20.0%
2008	812	211	20.6%			
2009	653	606	48.1%	17887	9270	34.1%
2010	735	195	21.0%		4054	100.0%
2011	13	344	96.4%			TBD
Average	735	571	45%	45553	14099	36%

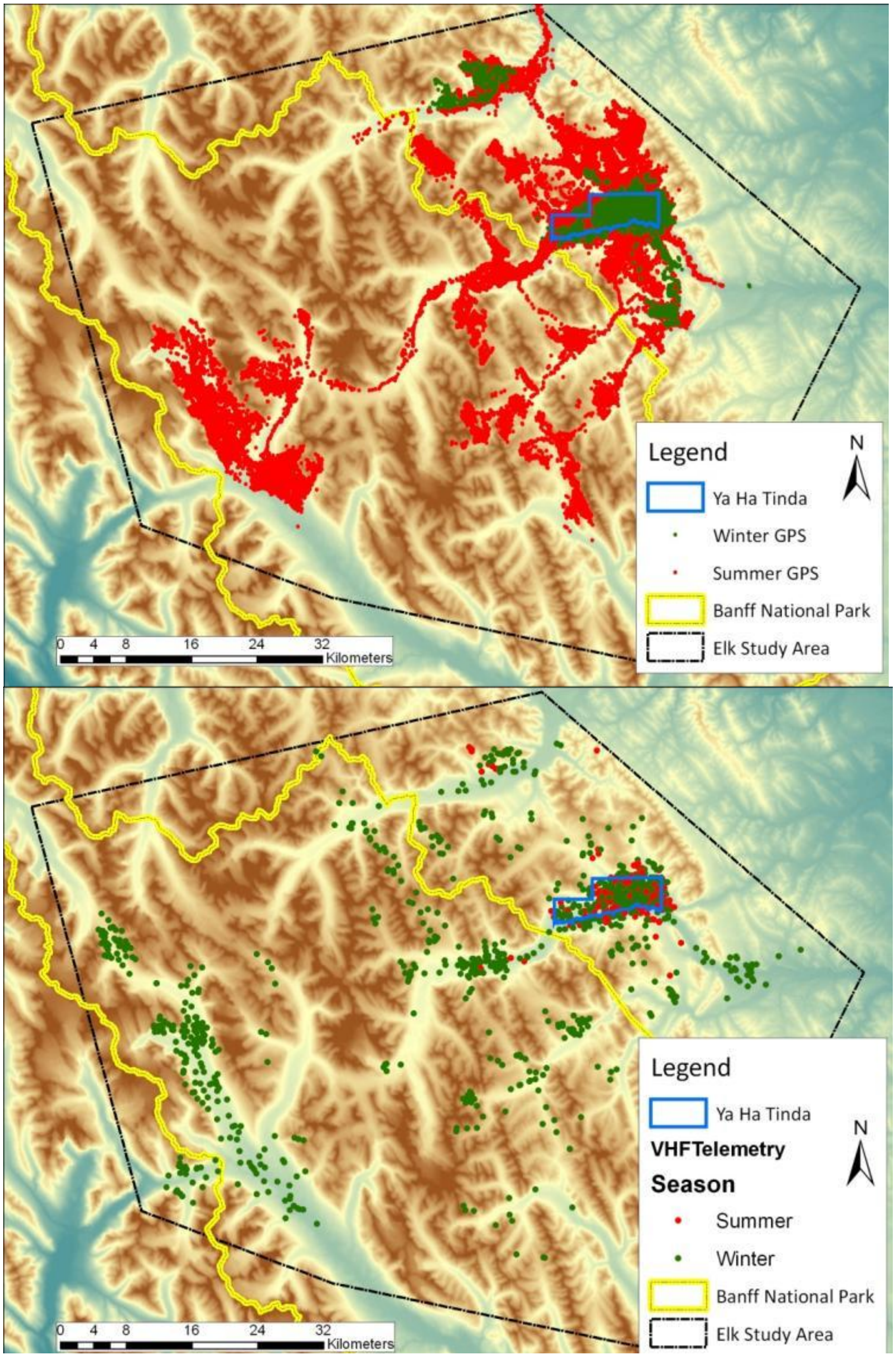


Fig. 8.2. VHF (bottom panel) and GPS (top panel) telemetry locations from 226 radio-collared adult female elk from 2001 - 2011, Ya Ha Tinda ranch and Banff National Park, Alberta, Canada.

Adult Female Elk Mortality Monitoring

As of 31 March 2011, we detected a total of 114 mortalities of radio-collared adult female elk. Wolves were responsible for 23, grizzly bears 11, cougars 7, disease 2, hunters 16, railway 1, unknown causes 46, and 9 were miscellaneous/other (Fig. 8.2a, b). On average, during 2002 - 2005, mortalities were investigated within 6.87 (SE = 1.91) days of mortality, and during 2006 - 2011, mortalities were investigated within 14.41 (SE = 2.85) days of mortality. Investigation times for 2006 - 2011 were significantly greater than during 2002 - 2005 (unequal variance t-test = -2.20, n = 81 known-aged kills, P = 0.03), suggesting a difference in monitoring intensity between the two periods may have influenced the determination of the mortalities causes. Future analyses will examine this in more detail.

Of all mortality since 2002, 37% was unknown, 20% was wolf, followed by 14% hunter, 10% grizzly bear, 6% cougar, and small amounts of other miscellaneous sources of mortality. Considering only known mortalities, wolf was 32%, hunter 22%, grizzly bear 15% and cougar 10%. Mortality was higher during late winter (Feb - Apr), summer (June - July), and during bow hunting and the late hunting season (Sept and Dec) (Fig. 8.4). Wolf mortality occurred year-round with a peak in June, as did grizzly bear predation, which also concentrated during June – September. Cougar mortality was greatest in the late winter (Fig. 8.4). To test if there was a difference in the frequency of mortalities between the two main periods (2002 - 2005 vs. 2006 - 2011) we compared proportions using Chi-square analysis, but found no differences in the frequencies between periods ($X^2_{,6}$ = 1.79, P = 0.31). There seemed to be no overall annual pattern in the timing of unknown mortalities (Fig. 8.4), indicating that our ability to ascribe cause-specific mortality was not biased over the course of the year. Mortalities were widely distributed (Fig. 8.5), but did not differ substantially from the distribution of collared elk (Fig. 8.3). Overall, the reported sources of mortality from the entire 2001 - 2011 study period seem not to differ from those reported during 2001 - 2005 by Hebblewhite and Merrill (2011).

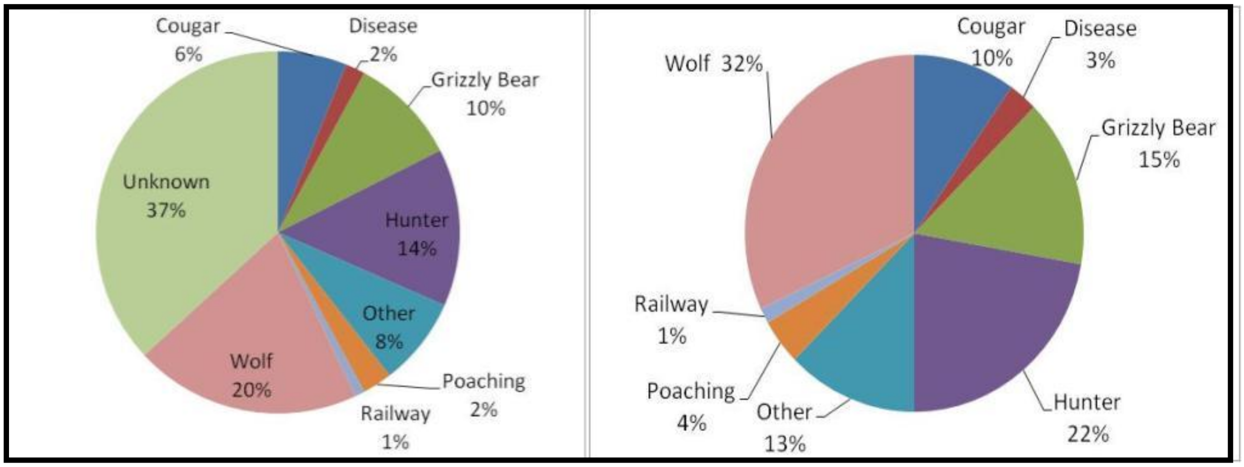


Fig. 8.3. Mortality causes for radio-collared adult female elk (n = 117) from 2001 - 2011 in the Ya Ha Tinda elk population, Alberta, Canada. (a) shows all mortalities, including unknowns, and (b) shows only known-causes of mortality excluding unknowns.

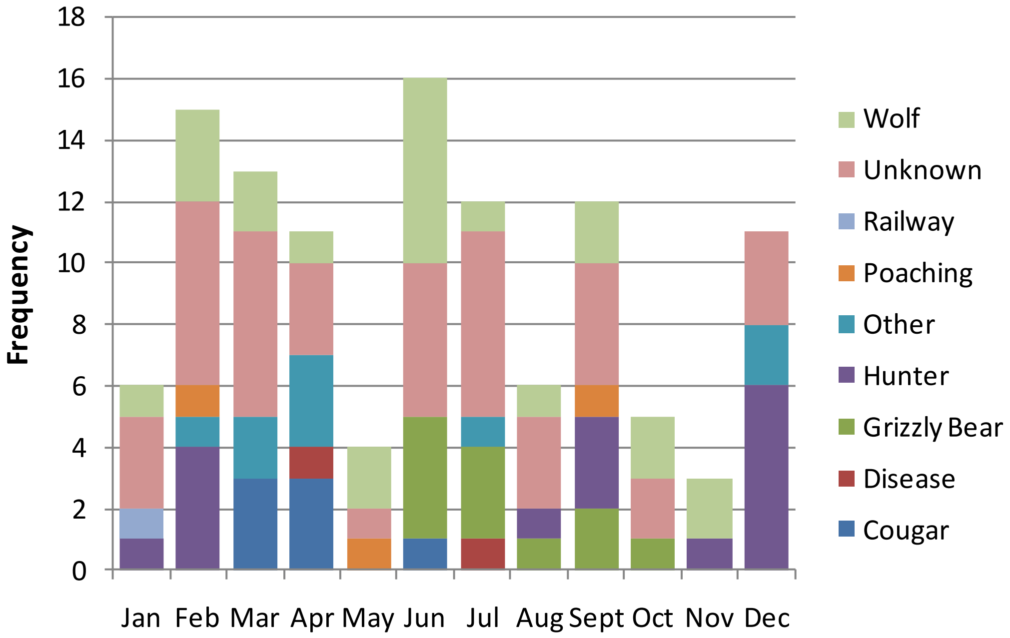


Fig. 8.4. Distribution of mortality causes over the calendar year for adult female elk in the Ya Ha Tinda elk population, 2001 - 2011, in Alberta, Canada. The dashed line shows the expected number of mortalities if mortality rate was constant over time.

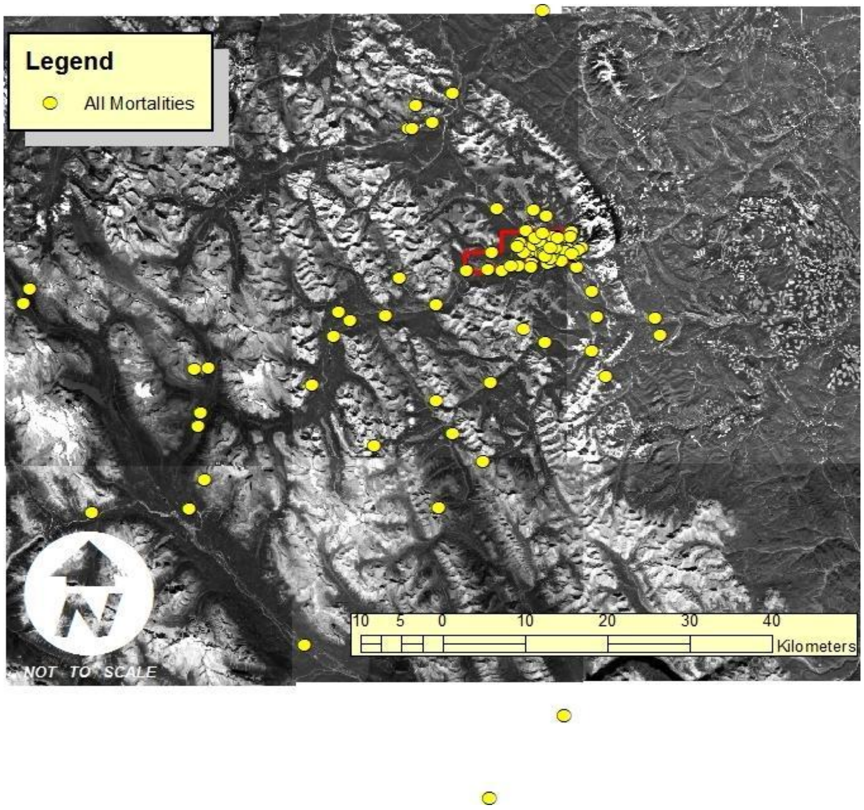


Fig. 8.5. Distribution of elk mortalities from 2002 - 2011, Ya Ha Tinda elk population, Alberta, Canada.

Age Distribution

We aged a total of 96 tooth samples collected from elk mortalities from 2005 to 2009 (*n* = 50) and from elk trapped in 2009 (*n* = 46). Elk age ranged from calves to 22 years. Of the dead elk, 4% (*n* = 2) were fawns, 32% (*n* = 16) were animals between 1 - 5 years of age, 30% (*n* = 15) were animals 6 - 10 years of age, 22% (*n* = 11) were 11 - 15 years of age, 10% (*n* = 5) were 16 - 20 years of age, and 2% (*n* = 1) were over 20 years of age. The age distribution of the captured elk was: 54.3% (n = 25) were 1 - 5 years of age, 23.9% (*n* = 11) were 6 - 10 years of age, 15.2% *(n* = 7) were 11 - 15 years of age and 6.5% (*n* = 3) were 16 - 20 years of age (Fig. 8.6). We tested for a difference between the distribution of age of captured elk and the ages from elk mortalities using a Kolmogorov-Smirnoff test. A significant difference would indicate a potential capture bias over time. However, the Kolmogorov-Smirnoff test revealed that capture and mortality ages were similar (*P* = 0.23), confirming little evidence for bias over time in ages of captured elk.

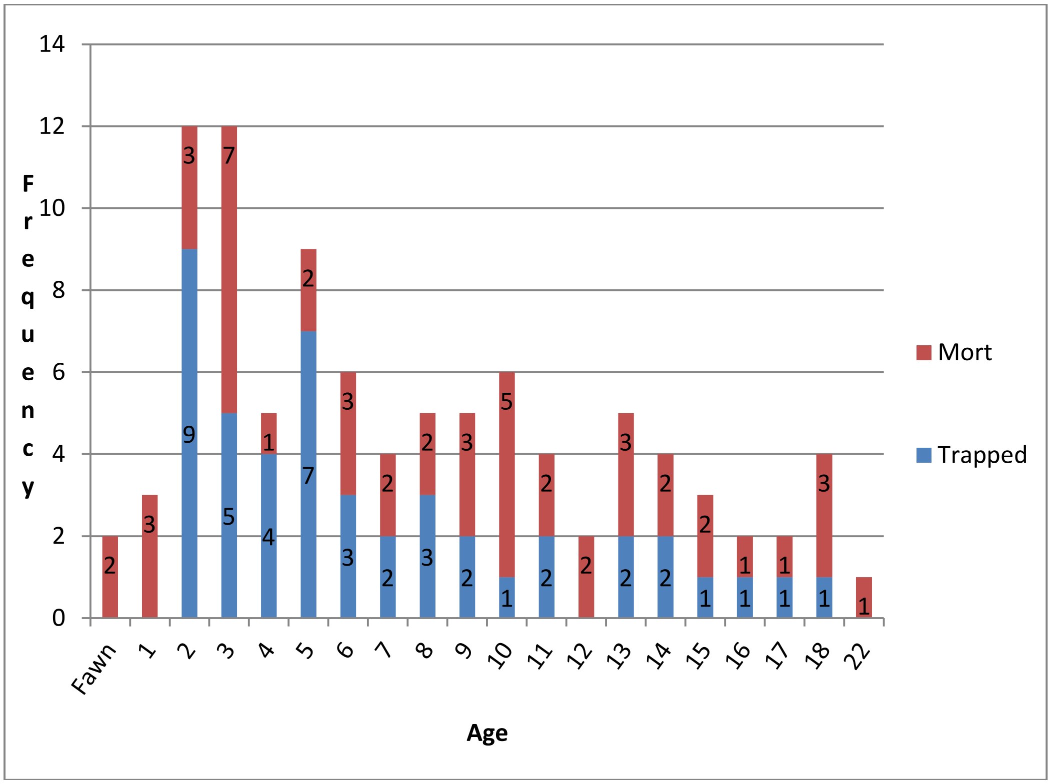


Fig. 8.6. Age distribution determined from tooth samples (collected 2005 – 2009) of elk found dead (red) and captured (blue) at the Ya Ha Tinda Ranch, Alberta, Canada.

Natality Rates and Cow:calf Recruitment

On average, 22 + SD elk groups of an average size of 102 + SD elk were classified each winter (except winter 2008) for an average of 2256/winter. Recruitment rates to late winter varied from a low of 0.09 in 2003 to 0.251 in 2006, and averaged 0.169 + SD from 2002 – 2011 (Table 8.6). There was a non-significant increasing trend in recruitment from 2002 – 2011 ($F_{1,9}$ = 2.82, P = 0.11, n = 9), shown in Fig. 8.7 that explained 22% of the variation in cow:calf ratio despite its weak statistical significance.

Table 8.6. Recruitment to late winter (Feb to Apr) based on cow:calf ratio data, Ya Ha Tinda elk herd, Alberta, Canada.

Year	Total Number Classified	# of Groups	ADF Total	YOY Total	Cow:calf
2002	1942	20	1361	186	0.137
2003	6296	70	5473	493	0.090
2004	4381	35	3563	533	0.150
2005	229	10	180	19	0.106
2006	2144	19	1385	347	0.251
2007	2316	14	1769	346	0.196
2008	--	--	--	--	--
2009	1568	13	316	86	0.272
2010	454	6	722	124	0.172
2011	977	11	360	55	0.153
	2256.3	22.0	1640.9	241.1	0.169

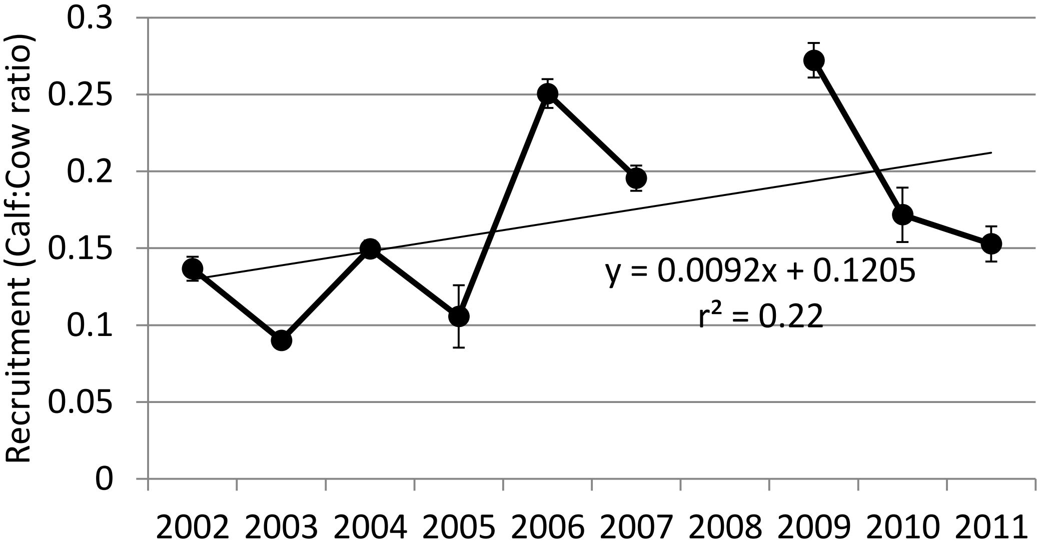


Fig. 8.7. Recruitment rates to late winter (Feb - Apr) and standard error from 2002 - 2011 based on cow-calf ratio data for the Ya Ha Tinda elk herd, Alberta, Canada.

Natality rates were higher than recruitment rates as expected, but not by much. On average 90 elk groups of 17 + SD elk were classified during each summer from June – August, for a total of 13,959 elk classified. Average natality rates were 0.242 with a range of 0.095 during 2007 (an obvious anomalous year) to 0.395 in 2006 (Table 8.7). There was an apparent large drop in natality rates observed during 2007 and 2008 (Fig. 8.8). The slight increasing trend in natality rates (Fig. 8.8) was not significant ($F_{1,9} = 1.22$, $P = 0.61$, n = 9).

Table 8.7 Natality (June – Aug) cow:calf ratio data, Ya Ha Tinda elk herd, Alberta, Canada.

Year	Sum of # Classified	Count of # Classified	Sum of ADF	Sum of YOY	Cow:calf ratio
2002	662	59	487	130	0.267
2003	1873	109	1454	372	0.256
2004	2012	105	1442	437	0.303
2005	598	32	422	111	0.263
2006	394	17	258	102	0.395
2007	734	37	600	57	0.095
2008	1367	55	1099	128	0.116
2009	2438	71	1709	526	0.308
2010	3883	321	2664	455	0.171
Average	1551	90	1126	258	0.242

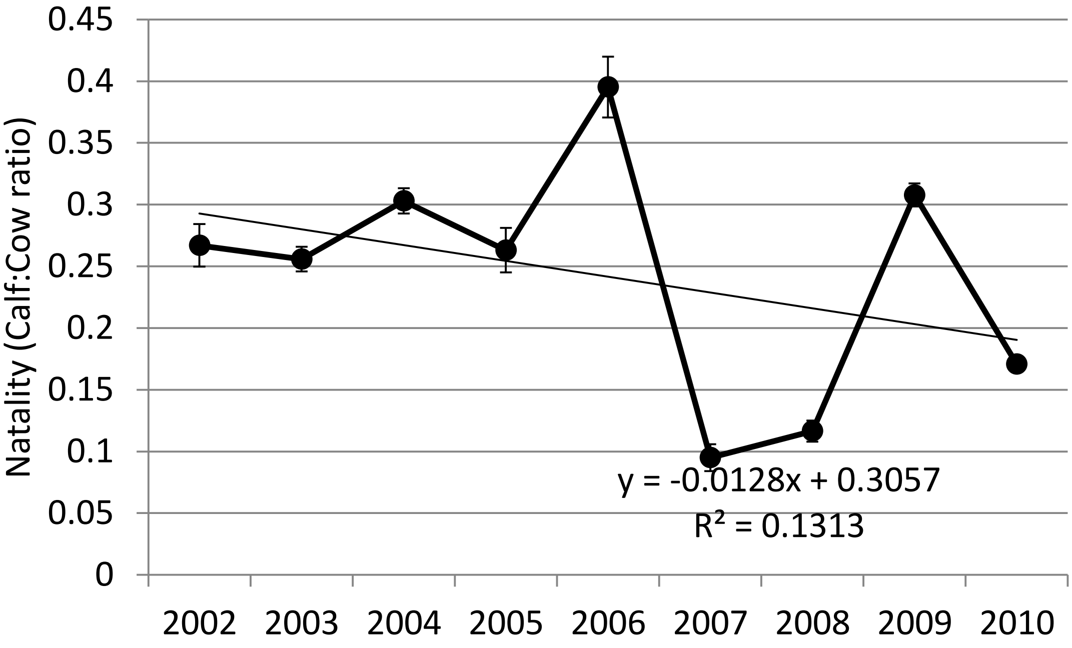


Fig. 8.8. Natality rates and standard error from 2002 - 2011 during summer (June – Aug) of the Ya Ha Tinda elk herd, Alberta, Canada.

Cow:calf recruitment and natality rates were comparable in mean and variation (Fig. 8.9) to historic data from 1972 to 2002 (Hebblewhite 2006, Table 8.10). In comparison to the entire period, the low natality rates of 2007 and 2008 do appear to be real declines compared to the long-term data, although the wide variation year-to-year suggests some sampling variation.

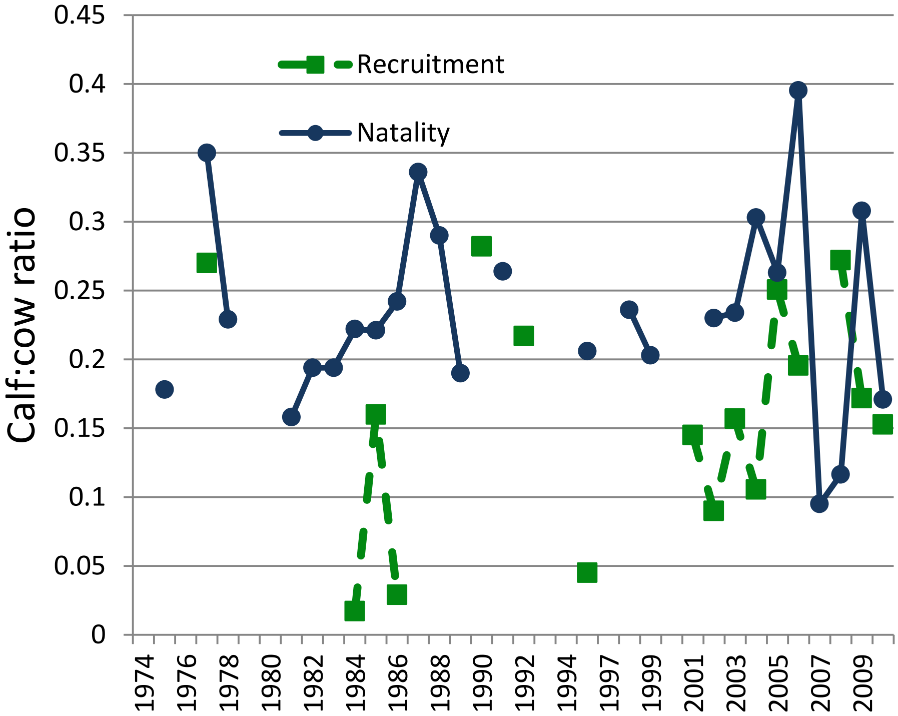


Fig. 8.9 Recruitment (cow:calf ratio during Feb – Apr) and natality rates (cow:calf ratio during June – Aug) from 1974 – 2011 for the Ya Ha Tinda elk herd, Alberta, Canada.

Winter Aerial Elk Surveys

Winter surveys revealed that the Ya Ha Tinda elk population continued to decline slowly from 2002 - 2011 from a population peak of 991 in bio-year 2002 to a low of 423 in 2010, compared against the slow long-term decline from a peak of 2089 in 1993. Graphing elk population size at time t versus t + 1 reveals that there may have been two stable points in this elk herd's trajectory, around 1000 elk from 1996 to 2005, and around 500 elk since 2005. Next, we followed the same methods as Hebblewhite et al. (2006) to test for carrying capacity with predation where $r_t = 0$, and found a long-term stable attractor of 589 elk (Fig. 8.12), although the relationship between r_t and N_t did not explain that much variation in elk population growth rate ($F_{1,31} = 3.21$, $P = 0.03$, $R^2 = 0.22$).

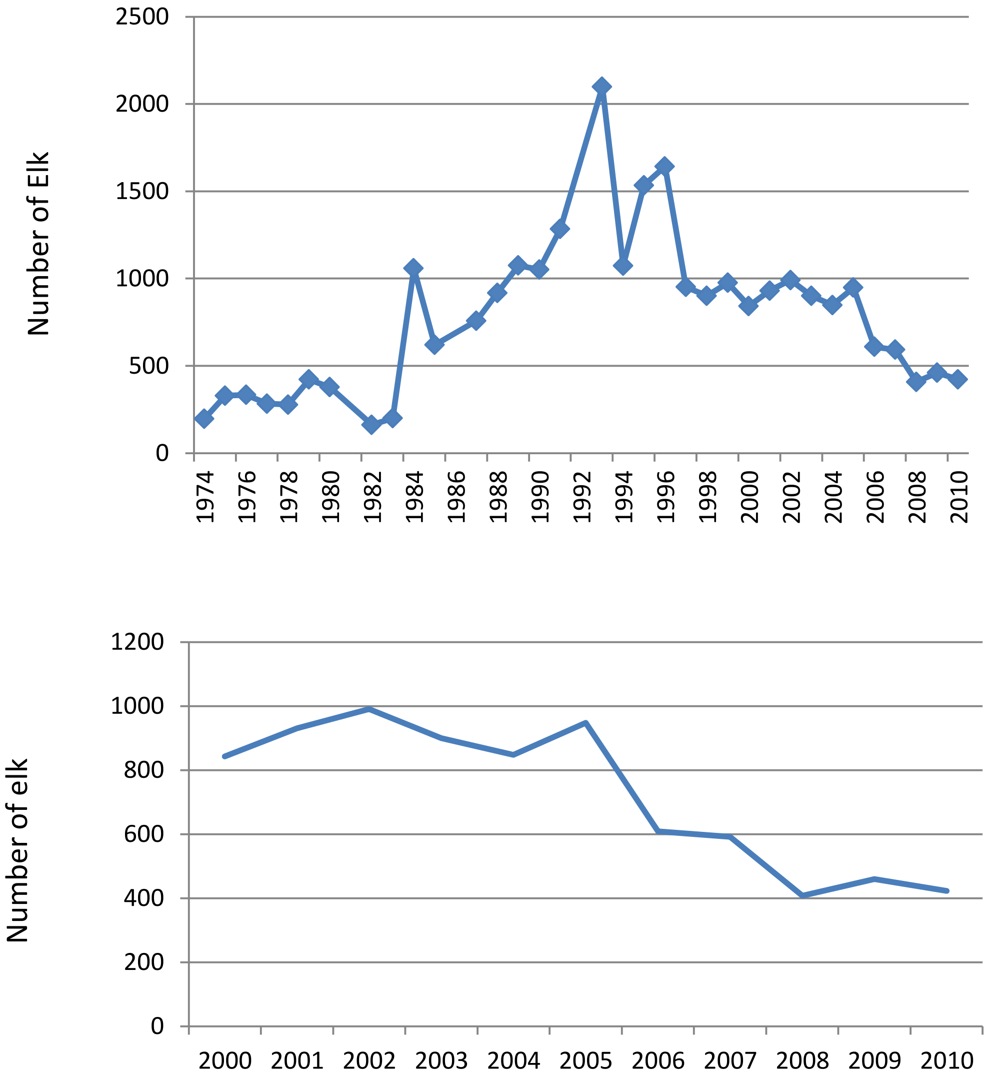


Fig. 8.10. Winter aerial elk counts for the Ya Ha Tinda winter range conducted by Alberta Fish and Wildlife and Parks Canada, with the top panel showing trends from 1974 - 2011, and the bottom panel focusing on trends since 2000 - 2011, Alberta, Canada.

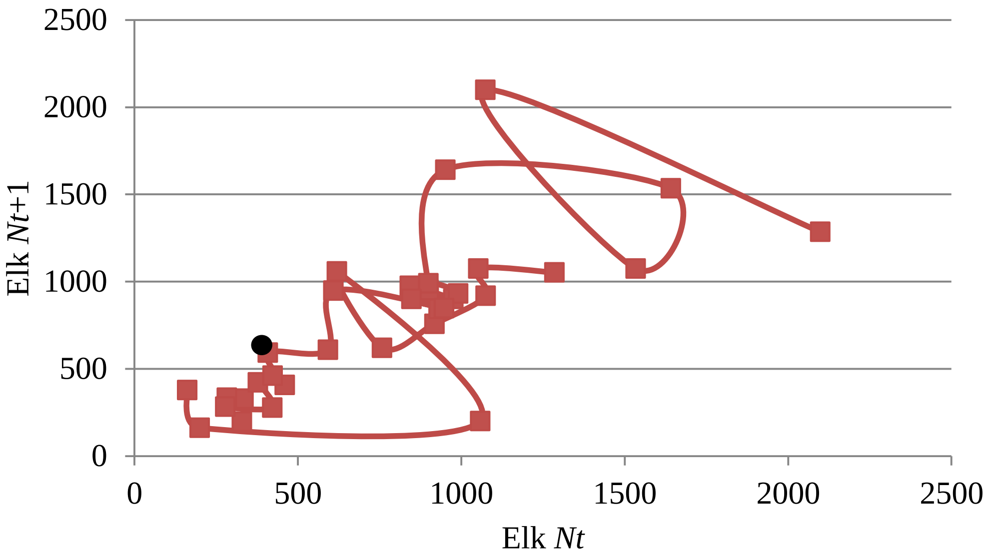


Fig. 8.11. Phase plane diagram of Ya Ha Tinda elk population size at time *t* versus *t* - 1. Phase plane diagrams help identify stable equilibria points, revealed by the two potential clusters of points at approximate population sizes of 1000 and 450. The black dot indicates the stable attractor identified in Fig. 8.12 by regressing population growth rate, r_t, versus population size.

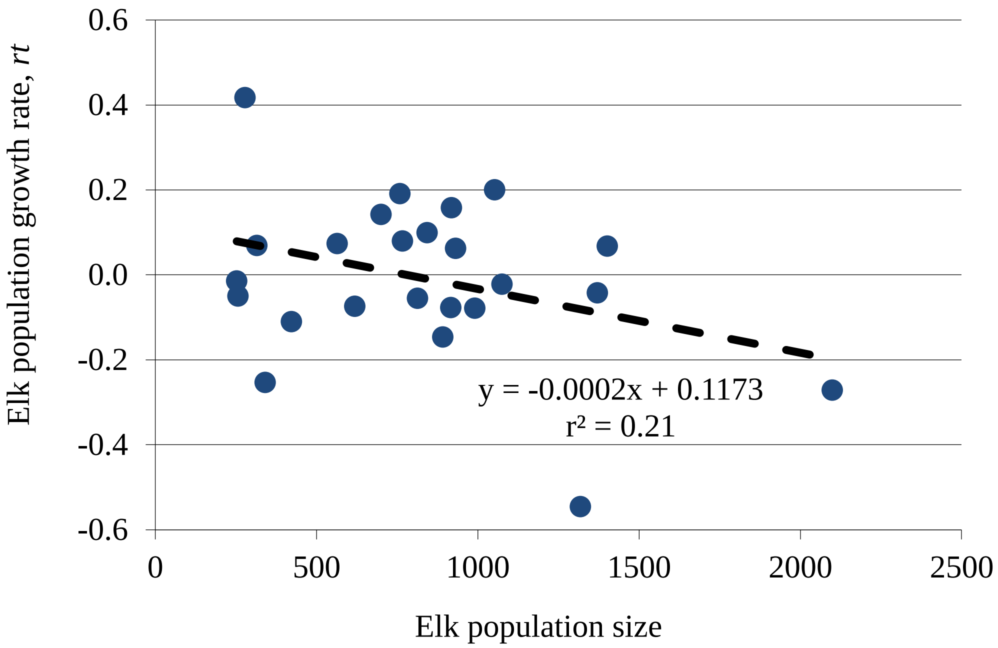


Fig. 8.12. Plot of population growth rate versus population size at time *t*, revealing the stable attractor where $r = 0$ at $N = 589$ elk. The regression is significant ($P = 0.03$), although variable.

Migratory Behavior

The proportion of migrants to residents in the Ya Ha Tinda elk herd has declined since the late 1980s at which point ~95% of the elk herd migrated to the high elevation summer range within the boundary of Banff National Park (Morgantini 1988, Hebblewhite et al. 2006). In 2001, the ratio of migrant to resident elk was estimated at over 5:1, or >80% were migrants. By 2010, the ratio of migrants to residents fell to an estimated 1:1, or 50% using comparisons of summer elk counts on the winter range as a function of the previous winter counts (Fig. 8.10).

One interesting shift in the migratory behavior of the Ya Ha Tinda elk herd has been the apparent increase in the number of elk migrating eastward towards the Dogrib burn and Mountain Aire Lodge. During the summer of 2010, a range of 20-25% of the marked adult female elk migrated east towards the Dogrib and Mountain Aire areas, in comparison to the westward migration of ~30% of the total radio-collared sample of elk (Fig. 813).

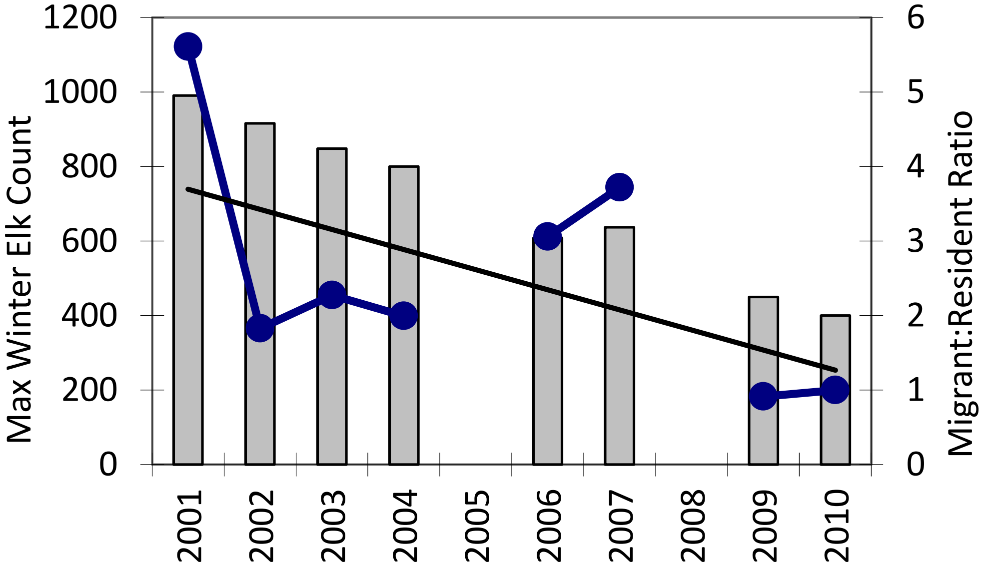


Fig. 8.13. Proportion of migrant to resident elk in the Ya Ha Tinda herd, Alberta, Canada, from 2001 - 2010 with maximum winter counts. The proportion is calculated from summer elk counts on the winter range as a proportion of the largest count from the previous winter.

Adult Female Elk Survival

Annual adult female survival rates for migrants and residents were estimated for each year using a non-parametric Kaplan-Meier estimator (Fig. 8.14). Mean annual survival rates for adult female elk were 0.904 (SE = 0.046). Average survival rates for migrants were slightly higher 0.910 (SE =0.048) than residents 0.899 (SE = 0.044) since 2002 (Fig. 8.15). Regressing survival estimates for resident and migrant elk against density in preliminary Cox proportional hazards regression models (S. Eggeman, University of Montana, unpublished data) shows that resident survival rates are density dependent, whereas migrant survival rates are not.

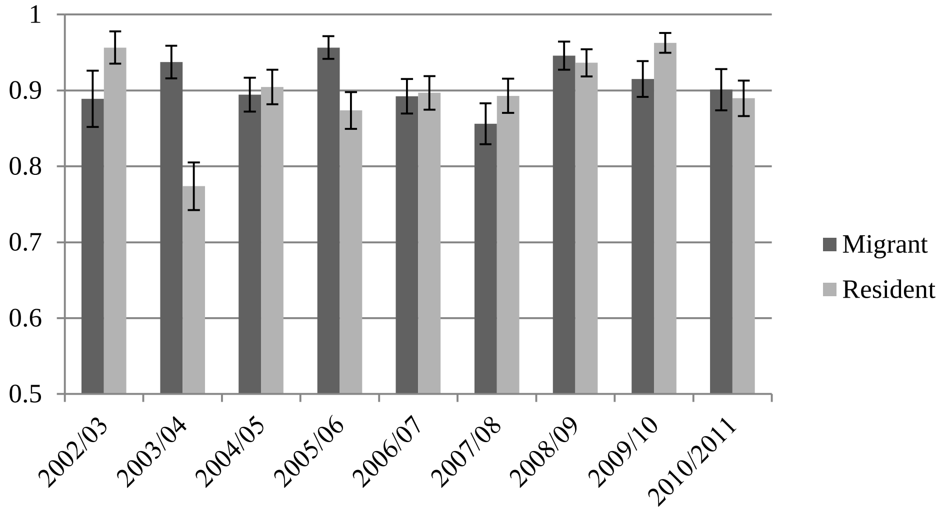


Fig. 8.14. Kaplan-Meier survival rates for the adult female elk in the Ya Ha Tinda elk population, Alberta, Canada, 2002-2011, comparing migrant and resident elk survival rates with standard errors estimated from KM survival modeling.

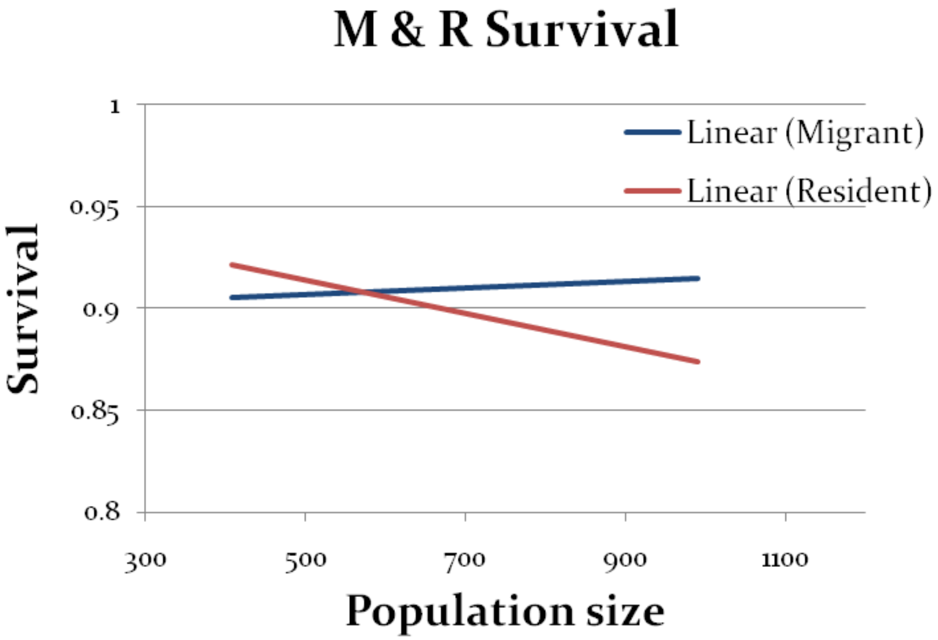


Fig. 8.15 Preliminary assessment of differential density dependence in adult female elk survival rates as a function of density from Cox proportional hazards regression in the Ya Ha Tinda elk herd, Alberta, Canada, 2002 - 2011. Results show that resident survival is density dependent, whereas migrant survival remained constant during the study.

We addressed the objective of understanding what factors drove elk population growth rates using life-stage simulation analysis (LSA). In 1000 random matrix model simulations, the coefficient of determination (r^2) between ƛ and vital rates was the highest for calf survival for migrants and residents, $r^2 = 0.834$ and 0.825, followed distantly by prime-age adult survival of $r^2 = 0.073$ and 0.05 (Fig. 8.16). All other LSA r^2 were < 0.05 (Hebblewhite 2006), indicating these 2 rates (calf survival and adult female survival) accounted for ~90% of the variance in λ. The maximum proportional change observed in ƛ from changing calf and adult survival rates was 0.30 and 0.12 for both strategies, indicating calf survival had a 2.5-fold greater maximum effect on λ.

Pregnancy Rates

A total of 285 pregnancy samples were collected from 2001 to 2011, with 229 of those positive for an average pregnancy rate of 0.804 and a range from 0.575 to 0.967 across all years (see Table 8.8, Figs. 8.17, 8.18). Migrants had significantly higher pregnancy rates of 88% compared to residents with 77% pregnancy rates (t-test *P*-value = 0.02). Pregnancy rates were variable in time as well, with high rates in bio-year 2004/05 and 2008/09, and the lowest pregnancy rate in 2007/08.

Table 8.8. Summary of winter pregnancy rates across all years except 2006/07 and 2009/2010 for the Ya Ha Tinda elk herd, Alberta, Canada.

Bio Year	# Pregnant	Total Samples	% Total
2001/02	23	35	0.657
2002/03	39	47	0.830
2003/04	41	49	0.837
2004/05	29	30	0.967
2005/06	20	26	0.769
2007/08	23	40	0.575
2008/09	40	42	0.952
2010/11	14	16	0.875
Total	**229**	**285**	**0.804**

8.4 Discussion

Overall, demographic evidence from aerial surveys, calf recruitment, adult female survival and pregnancy rates suggest that the Ya Ha Tinda elk herd may have reached a low-density equilibrium density of ~500 elk in 2005 that only continued monitoring will verify. The possible low-density equilibrium appears to be held there by high predation rates on adults by a diversity of predators. At the same time, migrant to resident ratios appear to have stabilized around 1:1 since 2005, indicating that about 50% of the population is migratory in 2011. Our recent results extend some of the findings of Hebblewhite and Merrill (2012) who studied elk demography from 2002 - 2004 and found that migrants and resident elk were both declining, although migrants about 2-3% faster than residents, largely due to higher predation rates from wolves during winter, grizzly bears during June (both inside and outside of BNP), human hunting in the fall (archery), and especially late winter hunting by First Nations hunters. Since 2003, there has been no archery season, and yet human harvest mortality continues to be high, mostly from First Nations hunters. The only other main difference between 2002 - 2004 and 2005 - 2011 appears to be reduced mortality from grizzly bears during the summer, and perhaps a slight reduction in wolf-caused mortality that might be related to increasing deer in the region.

These slight changes in cause-specific mortality appear to have driven a density-dependent increase in adult female survival from 2005 - 2011, especially for resident elk (Hebblewhite and Merrill 2011) who may be benefitting from reduced wolf and grizzly bear predation at lower densities as the population seems to be stabilizing around 500 total elk, or ~ 250 resident elk. However, despite the potential density-dependent declines in resident mortality, resident pregnancy rates were still much lower during 2001 - 2011 by an average of 10%, consistent with the reduced forage quality that residents obtained by foregoing migration to high forage quality summer ranges as shown by Hebblewhite et al. (2008). Therefore, it still appears that the same trade-offs that Hebblewhite and Merrill (2009) described between increased forage and high predation rates for migrants vs. low forage quality but reduced predation rates may still be in play, and only slightly benefitting residents through reduced mortality perhaps because of density-dependent declines in predation rates.

There also may have been slight density dependent increases in recruitment, as we found a slight, but not significant, increasing trend in Feb - April cow:calf ratios (Fig. 8.7), concurrent with declining density at the Ya Ha Tinda elk herd. Based on the preliminary evidence for a density-dependent (DD) response in resident elk survival rates, one might expect a similar DD response in resident calf survival. However, given our previous results that showed migrant elk benefitted more from higher quality forage in summer than did residents, which was associated with increased 8-month old calf weights, there could also have been DD increases in migrant calf survival. The apparent stability of the M:R ratio and possible population size (*N*) from 2005 - 2011 also supports the hypothesis that migrant elk have experienced increasing calf survival because otherwise, the increasing resident adult survival rates would have led to increasing ratio of residents in the population. Unfortunately, it is not possible to determine whether migrants or resident elk recruitment have differentially changed in response to reduced densities from cow:calf ratios alone. Following the mark-resight methods of Hebblewhite and Merrill (2011) from 2002 - 2004, our current thesis student Scott Eggeman is continuing mark-resight resident and migrant elk calf survival rates during summers 2010 and 2011 and we will be able to test for differences in strategy specific calf survival rates using these data. Either way, from 2002 - 2011, cow:calf ratios averaged 0.169, which, based on Harris et al. (2008), corresponds to a calf survival rate of about 0.21 (and a range of 0.269 - 0.40), which is higher than reported for 2003 and 2004 (Table 8.6) and the mark-resight estimates reported by Hebblewhite and Merrill (2011) for the 2002 - 2004 period.

Regardless of whether migrant or resident (or both) calf survival has increased, they may be getting to high enough levels for stable or even increasing population growth rates (Raithel et al. 2007, Harris et al. 2008). Our life-stage simulation analysis supports that it is likely changes in calf survival that are driving long-term changes in elk population growth rate (Figure 8.14), and future studies on elk calf survival may help elucidate differences between resident and migrant elk.

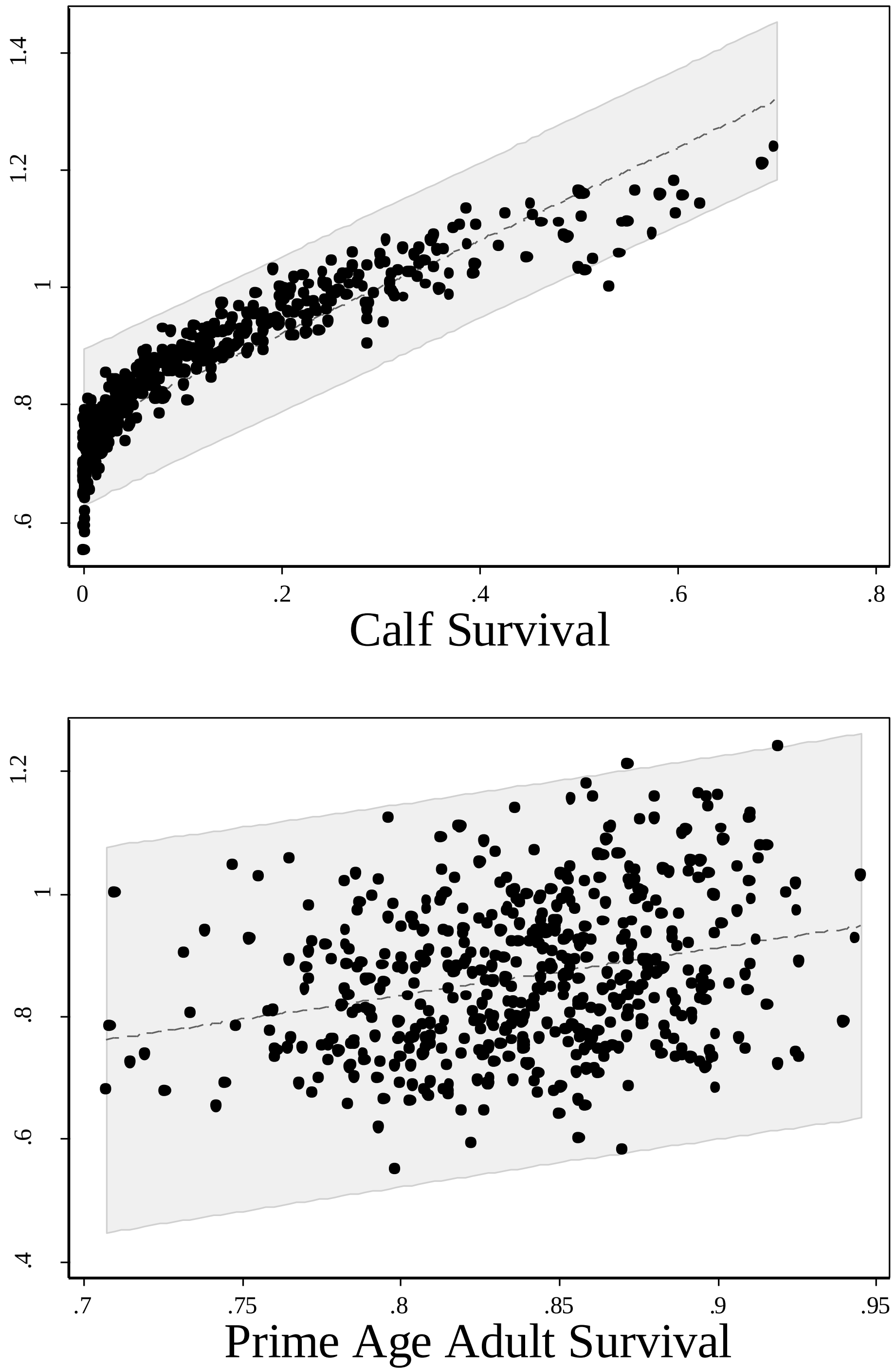


Fig. 8.16. Population growth rate (finite, lambda λ) of migrant elk as a function of the 2 most important vital rates, a) calf survival and b) prime-aged adult female survival, for explaining variance in Ya Ha Tinda elk population growth from 500 simulated matrix models from life-stage sensitivity analysis based on within-study process variance in vital rates (Raithel et al. 2007).

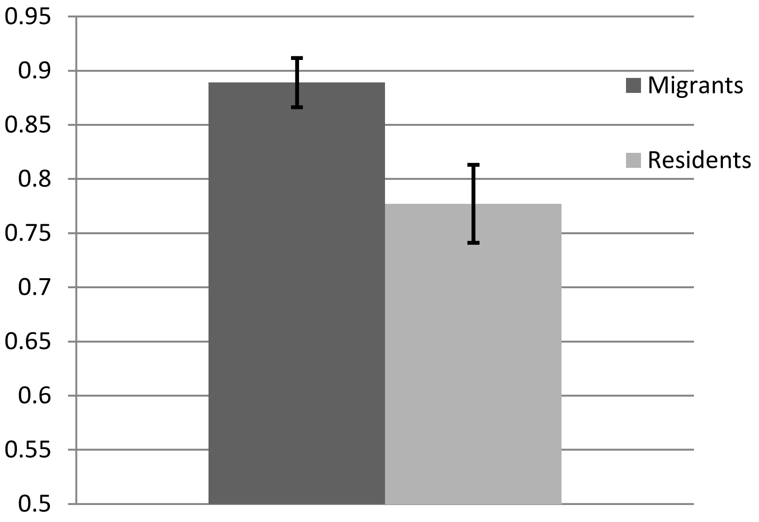


Fig. 8.17. Mean pregnancy rates for migrant and resident elk in the Ya Ha Tinda elk population from bio-year 2001/02 – 2010/11, Alberta, Canada.

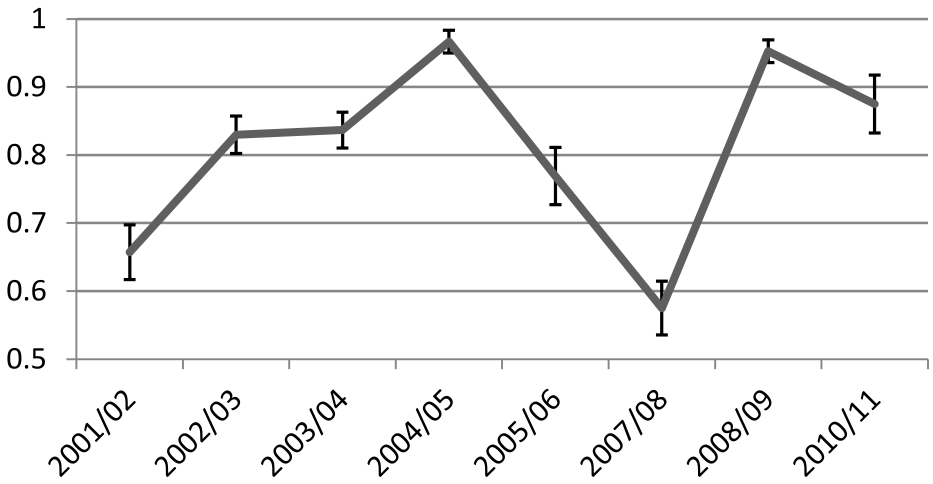


Fig. 8.18. Pregnancy rates (and SEs) from bio-year 2001/02 – 2010/11 for the Ya Ha Tinda elk herd, Alberta, Canada.

9.0 Summary and Conclusions

- Park managers and ecologists are concerned that the current state of the montane ecoregion of BNP is outside the historical range of variability for the area (Achuff et al. 1996). Montane grasslands in the RDRV are an important vegetative component for many wildlife species including ungulates. Long-term fire cycles in montane grasslands range from <50 years but since the 1950s fires have been less frequent. A consequence of fire exclusion has resulted in a steady encroachment of conifers and shrublands that reduce both forage quantity and quality for large herbivores except where human manipulations such as mowing and prescribed burning exist.

- While prescribed burning at SC and TY has occurred to maintain grassland communities, fires have not been sufficient to arrest conifer encroachment (at least compared to the 1950s) in the vicinity of the grassland. At YHT and WL, it has been less feasible to burn the grasslands due to fuel reductions because of high elk populations and in the recent past, drought conditions. Instead, mechanical mowing has maintained or slightly increased the extent of grasslands at YHT, but not at WL, where conifer encroachment has been the greatest. Recent burn programs at YHT and WL have removed tree and shrub cover, and with the recovery of grassland litter, fires have spread into portions of the grasslands as well.

- Long-term monitoring of grassland vegetation has provided information on annual production and changes in grassland production with changing precipitation and ungulate numbers and distribution. Further analysis of these relationships is expected in winter 2012.

- The collaborative demographic study at YHT is one of the longest running and largest long-term studies of elk, especially in high predation systems. Since 2002, we have monitored over 226 individual adult female elk for resource selection, migration, and cause-specific survival rates, resulting in one of the longest radio-collared elk studies in North America. We have collected > 13,000 VHF locations from 192 different elk, and > 400,000 GPS locations from 56 individuals since 2002. Through radio-telemetry monitoring, we have identified a total of 114 mortalities, of which 63% were known causes of mortality. To estimate recruitment and natality, we have classified over 20,000 individual elk during the late winter and spring seasons, and have partnered with Alberta Fish and Wildlife to conduct aerial surveys of elk since 1974.

- Together, with six different graduate students, we have used these data to produce 24 scientific publications, 28 conference presentations, and leveraged Parks Canada funding by an average ratio of 2:1 with outside partners including Alberta Conservation Association, NSERC, ACA Biodiversity grants, etc.

- Analyses of long-term changes in elk distribution in response to aversive conditioning suggest that during the reduced intensity of aversive conditioning during 2008-2010, elk distribution began to revert to pre-conditioning distribution (pre-2005). However, analyses were confounded because of differences in methods over the duration of the study, and because GPS data is still stored on-board elk collars that were deployed during the 2009-

2011 period. Subsequent analysis using the GPS data of elk responses to the cessation of aversive conditioning in summer of 2011 will provide a definitive test of whether the elk will increase use of the main Ya Ha Tinda grasslands now that aversive conditioning is no longer taking place.

- Long-term pellet monitoring revealed that elk responded to thinning and burning treatments at the Ya Ha Tinda favorably in both winter and summer. During summer, elk increased their use and selection for burned grasslands especially, and by the second year post-burn, showed increased use of both burned grasslands, and thinned and burned areas. In contrast, deer (undetermined white-tailed or mule deer) seemed to avoid thinned and burned areas during winter and summer, but selected burned grasslands strongly during summer.

- Long-term demographic data reveal a population that appears to be stabilizing at a dynamic equilibrium of ~ 500 elk (50% of which are migratory and 50% of which are resident). Since 2005, there appear to have been slight density-dependent increases in adult female survival (especially of residents) and calf recruitment, which help explain this potential stabilization. Such density-dependent declines in mortality with decreasing density is consistent with a declining carnivore population (e.g., wolves) or declining predation as predators switch to alternate prey (e.g., white-tailed deer).

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Publications

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2. Hebblewhite, M., Merrill, E. H., Morgantini, L. E., White, C. A., Allen, J. R., Bruns, E., Thurston, L. & Hurd, T. E. 2006. Is the migratory behavior of montane elk herds in peril? The case of Alberta’s Ya Ha Tinda elk herd. *Wildlife Society Bulletin* 34: 1280-1295.
3. Hebblewhite, M. 2006. *Linking predation risk and forage to ungulate population dynamics*. Dissertation. Edmonton, Alberta: University of Alberta.
4. Hebblewhite, M. 2007. Predator-prey management in the National park context: lessons from a transboundary wolf-elk, moose, and caribou system. *Transactions of the North American Wildlife Conference* 72: 348-365.
5. Hebblewhite, M. & Merrill, E. H. 2007. Multiscale wolf predation risk for elk: does migration reduce risk? *Oecologia* 152: 377-387.
6. Hebblewhite, M., Percy, M. & Merrill, E. H. 2007. Are all GPS collars created equal? Correcting habitat-induced bias using three brands in the Central Canadian Rockies. *Journal of Wildlife Management* 71: 2026-2033.
7. Hebblewhite, M. 2008. Disentangling bottom-up and top-down effects of fires on montane ungulates. Pages 33-35 *in* R. E. Masters, K. E. M. Galley & D. G. Despain, editors. The '88 fires: Yellowstone and beyond. International Association for Wildland Fire Vol. 16, Jackson Hole, Wymoning, USA.
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14. Hebblewhite, M. & Smith, D. W. 2009. Wolf community ecology: ecosystem effects of recovering wolves in Banff and Yellowstone National Parks. *in* M. Musiani, L. Boitaini & P. C. Paquet, editors. The world of wolves: new perspectives on ecology, behavior, and policy. University of Calgary Press, Calgary, Alberta, Canada.
15. Robinson, B. G., Hebblewhite, M. & Merrill, E. H. 2010. Are migrant and resident elk (*Cervus elaphus*) exposed to similar forage and predation risk on their sympatric winter range? *Oecologia* 164:265-275.
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1. Hebblewhite, M., & Merrill, E. H. Ecology of the Ya Ha Tinda Elk Herd. Poster. 2003 Alberta Chapter of the Wildlife Society, Red Deer, AB.
2. Hebblewhite, M., & Merrill, E. H. Applying resource selection functions to modeling predator-prey dynamics. 2003 Alberta Chapter of the Wildlife Society, Red Deer, AB.
3. Hebblewhite, M. Synthesizing advances in predator-prey and ungulate-plant dynamics: what links are missing? Special symposium summary discussion, presentation. 2004 The Wildlife Society, Calgary, AB.
4. Merrill, E. H., Haydon, D. T., Frair, J., Hebblewhite, M., Visscher, D., Webb, N., & Friar, J. Landscape patterns and movement strategies alter wolf-elk encounter rates. Presentation. 2004 The Wildlife Society, Calgary, AB.
5. Hebblewhite, M., & Merrill, E. H. Modeling trade-offs between wolf predation risk and forage biomass using resource selection functions. Oral paper. 2004 The Wildlife Society, Calgary, AB.
6. Thurston, L., Hebblewhite, M., Morgantini, L.E., Merrill, E. H., & White, C. E. Elk population dynamics across 25-years of wolf recolonization in Banff National Park's northern range: The Ya Ha Tinda. Oral presentation. 2004 The Interagency Wolf Conference, Chico Hot Springs, MT.
7. Webb, N., Hebblewhite, M., Thiessen, C., & Merrill, E. H. Determining reproductive rates of wolves using fecal hormones. Oral paper. 2004 The Wildlife Society Conference, Calgary, AB.
8. Hebblewhite, M., Merrill, E. H., Morgantini, L. M., White, C., Thurston, L., Allen, J. R., Bruns, E., & Hurd, T. E. 2005. Is the migratory behavior of montane elk herds in peril? The case of the Ya Ha Tinda elk herd. Paper presented. 2005 Alberta Chapter of the Wildlife Society Conference, Red Deer, AB.
9. Munro, R. M., Hebblewhite, M., Visscher, D., Hamilton, S., & Merrill, E. H. Preliminary evaluation of elk forage dynamics after harvest and fire in the central east slopes of Alberta. Poster presentation. 2005 Alberta Conservation Association Partners in Conservation Conference, Calgary, AB.
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17. Merrill, E. H. Predator-prey dynamics in the Canadian Rockies. 2009 University of Calgary, Dept. Biological Science Seminar Series, Calgary, AB.
18. Robinson, B. W., & Merrill, E. H. Exposure of migrant and resident elk to forage and wolf predation risk at Ya Ha Tinda winter range: vigilance and foraging. 2009 Alberta Chapter of The Wildlife Society Annual Meeting, Edmonton, AB.
19. Merrill, E. H., Robinson, B. W., & Hebblewhite, M. Landscapes of fear: Wolf-elk behavioral responses in the Canadian Rocky Mountains. 2010 University of Washington, Seattle, WA.
20. Glines, L., Merrill, E., &White, C. Implications of a changing montane landscape: assessing the capacity for the YHT elk herd. 2010 Alberta Chapter of The Wildlife Society Annual Meeting, Red Deer, AB.
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22. Eggeman, S., Hebblewhite, M. & Merrill, E. H. Investigating the effects of top-down regulation on life-history strategy of a partially migratory elk herd. 2011 9th Western Deer and Elk Workshop, Albuquerque, NM.
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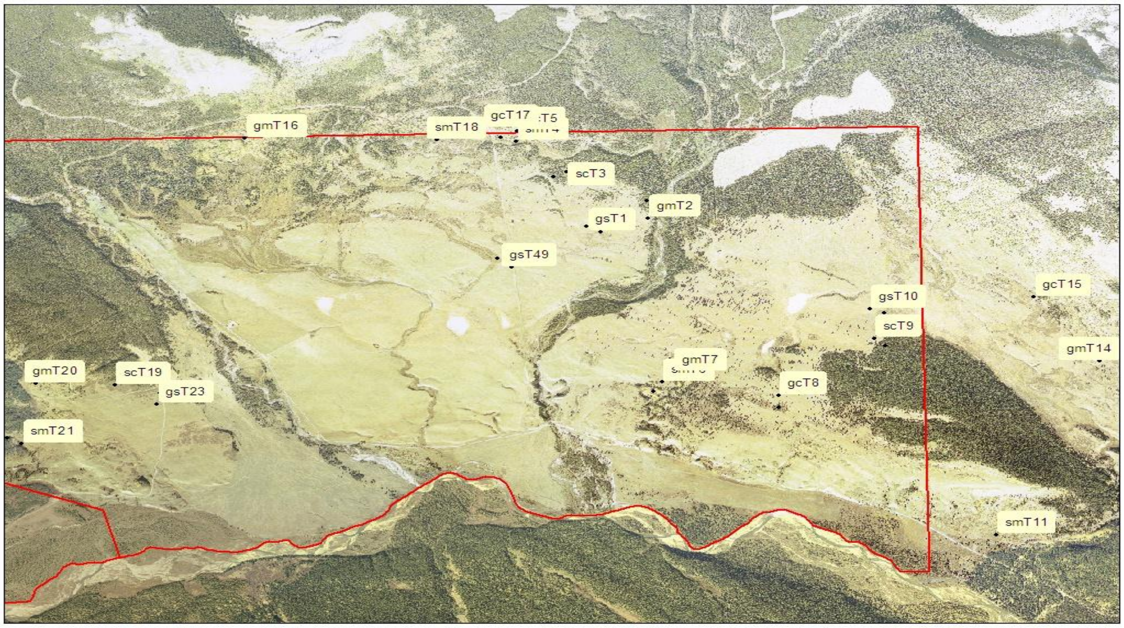
Posters

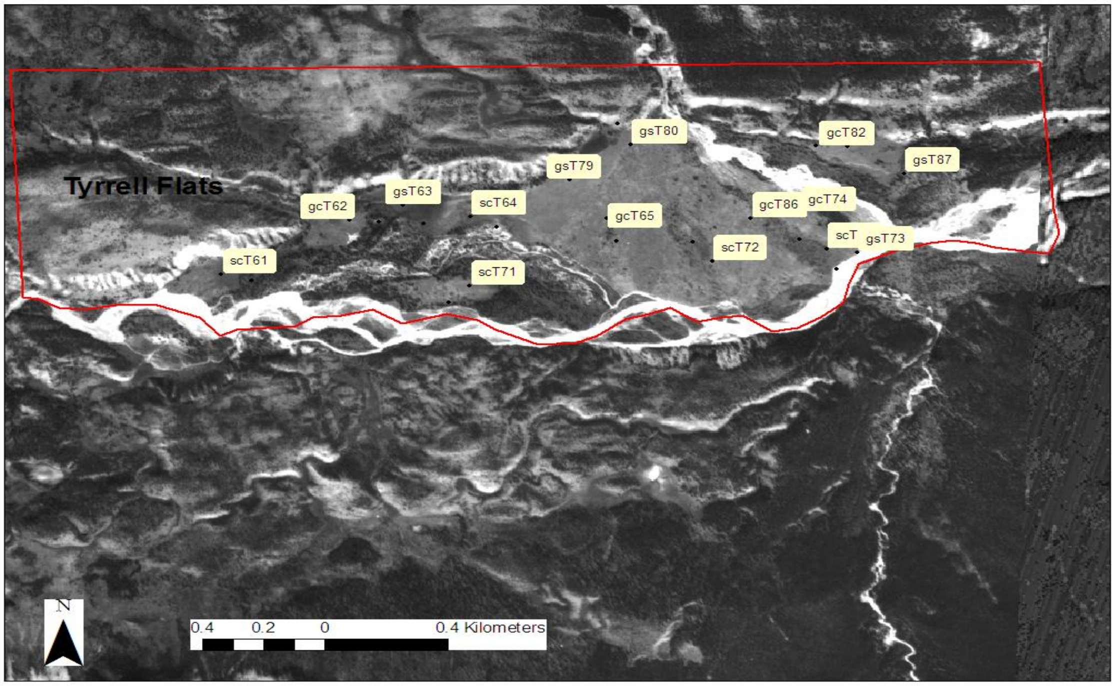
1. Glines, L., Merrill, E., & White, C. Grassland dynamics: investigating cause and consequences of woody plant encroachment. 2009 Alberta Chapter of The Wildlife Society, Edmonton, AB.
2. Glines, L., & Merrill, E. Implications of a Changing Montane Landscape: Assessing Nutritional Carrying Capacity for Elk of the Red Deer River Valley. 2010 International Congress for Conservation Biology, Edmonton, AB.
3. Glines, L. M., Merrill, E. H., & White, C. Shifting Landscapes: 50 years of grassland change in the montane region of the upper Red Deer River, Alberta. R.E. Peters Student Conference, University of Alberta, Edmonton, AB.
4. Glines, L. M., Merrill, E. H., & White, C. Shifting Landscapes: 50 years of grassland change in the montane region of the upper Red Deer River, Alberta. 2011 Canadian Society for Ecology and Evolution Conference, Banff, AB.

12.0 Appendices

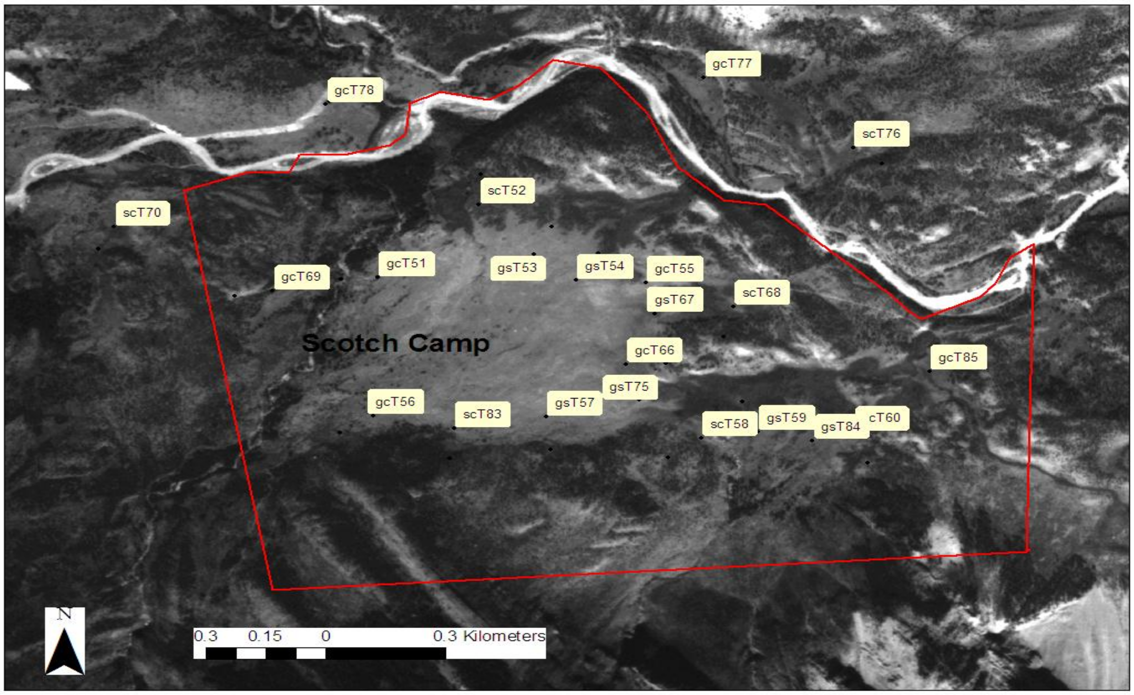


Appendix 1a. Transect locations within the Ya Ha Tinda ranch, Alberta, Canada, used to ground truth classified 2003 cover map.

Appendix 1b. Transect locations within West Lakes, Alberta, Canada, used to ground truth classified 2003 cover map.



Appendix 1c. Tyrrell Flats, Alberta, Canada: transect locations used to collect biomass and species data.



Appendix 1d. Scotch Camp, Alberta, Canada: transect locations used to collect biomass and species data.

Appendix II. Fire occurrence in the upper Red Deer River Valley, Alberta, Canada. Reconstructed from fire scar and stand origin studies (White et al. 1985, White et al. 2001). Marked (*) years indicated prescribed burns

SC	TY	WL	YHT
2005*	2005*	2009*	2009*
1999*	1994*	1929	2007*
1994*	1981*	1910	1936
1981*	1921	1899	1929
1931	1881	1889	1910
1921	1875	1879	1899
1879	1859	1864	1889
1859	1700	1846	1879
1849	1675	1839	1864
1831	1660	1817	1846
1808		1700	1839
1795		1645	1817
1778			1787
1771			1779
1754			1769
1720			1754
1700			1730
1627			1700

Appendix III. Classification types used to classify aerial photographs to study vegetation change through time in 4 study sites within the upper Red Deer River Valley, Alberta, Canada.

Classification Type	**Description
Grassland (≥ 60% grass cover)	Typically circular; size ranges with few large, many small patches Light tones (varies between images), but among the lightest; only bare ground or rock is lighter. No shadows cast Smooth texture Associated with low elevations
Shrubland (≥70% shrub cover)	Moderate patch size, typically smaller then coniferous patches Moderate height discernible when next to grassland or forest patches Medium tone, between grassland and coniferous Shadows not distinctive Fine texture; slightly ruffled Associated with low-lying wet areas, also with coniferous forest edge
Mixed-wood (≥ 60% deciduous cover)	Uniform, circular, Patches smaller then shrubland typically 'Salt and pepper' tone due to mix of deciduous (light) and coniferous(dark) species Shading leads to cloudlike texture of canopy Bottom or sides of slopes
Coniferous (≤>60% conifer cover)	Matrix. Edges may be convoluted or gradually blend; especially sharp where logged Often tell height due to shadow Darkest vegetation tones, only water & shadow is darker Slight texture of zigzag waves where canopy peaks and shadows fall Usually higher in elevation, bordering grasslands
***Logged**	Formerly coniferous in a previous time period, now bare (see road & bare ground) Only logged patch in 1982, in YHT and present in subsequent years
***Pond (water)**	Circular . Small (>8 ha each) Darkest tones Smooth texture in grassland areas (typically)
***River (water)**	Thin corridors running west to east and north to south Sinuous, braided pattern Wider then roads Darkest tones Associated with bare ground (gravel) from former channels
***Building**	Rectangular. Very small (>.35 ha each) Tone is very light (similar to bare) Shadows not visible Associated with each other and pasture
***Road & bare ground**	Bare patches smaller then shrubland patches, usually longer then wide Roads long and narrow; narrower and straighter then rivers Lightest tones Roads along valley bottoms highest peaks and area of sudden elevation change are bare
***Pasture**	Areas reserved for horse grazing and recently made inaccessible to elk/deer/wild ungulates with fencing Stable over time; appear the same as grassland except tone similar to shrublands

* grouped into 'Other' category for spatial extent summary
** General principles of air photo interpretation: Shape, pattern, size, tone, shadow, texture, and association.

Appendix IV. Reference points for comparison with 2003 photo set vegetation classification. Reference points were obtained from start and end points along 100-m transects spanning several ecotone types. Ecotones either began or ended in the following types depending on letter order, respectively: (g)rassland, (s)hrubland, (m)ixed-wood, or (c)oniferous. Transects were also labeled in order sampled, e.g., T1, T2…

Transect	Start UTM x	Start UTM y	End UTM x	End UTM y
gsT1	601009	5734371	601098	5734324
gmT2	601392	5734444	601388	5734597
scT3	600881	5734844	600795	5734800
smT4	600563	5735112	600467	5735143
gcT5	600566	5735202	600583	5735296
smT6	601485	5733022	601433	5732943
gmT7	601556	5733111	601534	5733218
gcT8	602223	5732906	602223	5732802
scT9	602824	5733402	602895	5733334
gsT10	602797	5733657	602888	5733619
smT11	603595	5731699	603666	5731762
gsT12	604688	5731985	604588	5731996
scT13	605090	5731998	605184	5732021
gmT14	604253	5733205	604335	5733257
gcT15	603836	5733758	603920	5733812
gmT16	598848	5735138	598908	5735209
gcT17	600349	5735225	600275	5735285
smT18	600064	5735121	600058	5735220
scT19	598024	5732994	598051	5733095
gmT20	597525	5733008	597539	5733091
smT21	597436	5732489	597346	5732538
gcT22	596668	5732175	596596	5732108
gsT23	598292	5732832	598308	5732927
gcT24	597034	5731402	596939	5731358
gcT25	593326	5730899	593232	5730895
scT26	593573	5731120	593657	5731166
gcT27	593234	5731503	593237	5731616
gcT29	594081	5730732	594051	5730830
gcT30	593897	5731302	593954	5731227
scT31	593642	5731456	593690	5731372
gcT32	593633	5730829	593576	5730900
gcT33	592321	5730611	592425	5730611
gcT34	592169	5730498	592067	5730493
gcT35	592917	5730790	593009	5730818
gmT36	592397	5731145	592393	5731245
gmT37	592209	5731076	592190	5731181
gmT38	591800	5731085	591845	5731178
gmT39	591383	5731213	591480	5731229
gmT40	591062	5731363	590964	5731375
scT41	591890	5730535	591908	5730436
scT42	591867	5730301	591858	5730404

gsT43	591748	5730908	591722	5730829
gsT44	591614	5730969	591528	5730991
gsT45	591800	5731085	591744	5731172
scT46	592534	5731257	592556	5731346
gsT47	592075	5730960	591980	5730963
gsT48	591769	5730831	591838	5730767
gsT49	600535	5734018	600443	5734092
scT50	600234	5735167	600142	5735181

Appendix V. Change in spatial extent of vegetation types (ha) in 4 study sites within the Red Deer River Valley, Alberta, Canada, from 1952 - 1992 for SC and TY, and 1952 - 2003 for WL and YHT.

Area of Vegetation Types (ha)

Site	Class	1952	1952 to 1962	1962	1962 to 1972	1972	1972 to 1982	1982	1982 to 1992	1992	1992 to 2003	2003	Total	
SC	Grassland	87.2	-14.9	72.3	-10.1	62.2	-3.5	58.6	-1.3	57.3	*		-29.8	(-34.2%)
	Shrubland	37.2	0.8	38.0	-7.2	30.8	-0.8	30.0	-1.7	28.2			-9.0	(-24.2%)
	Coniferous	119.0	6.7	125.7	24.3	149.9	2.5	152.4	2.8	155.2			36.2	(30.4%)
TY	Grassland	136.8	-24.7	112.1	-17.6	94.5	-10.0	84.5	-15.3	69.2			-67.7	(-49.4%)
	Shrubland	14.9	-3.5	11.4	1.9	13.2	0.7	13.9	0.0	13.9			-1.0	(-6.8%)
	Coniferous	142.5	17.0	159.4	16.0	175.4	4.1	179.5	12.8	192.3			49.9	(35.0%)
WL	Grassland	662.7	-43.4	619.3	-144.0	475.3	-179.3	296.0	-18.2	277.7	-103.3	174.4	-488.3	(-73.7%)
	Shrubland	33.8	-1.6	32.2	11.5	43.7	90.9	134.6	-70.0	64.7	31.7	96.3	62.5	(184.9%)
	Mixed-wood	11.3	3.1	14.4	7.0	21.5	-0.8	20.7	-5.5	15.1	6.7	21.8	10.5	(31.8%)
	Coniferous	694.2	29.2	723.4	123.6	847.0	90.1	937.1	106.3	1043.4	75.4	1118.7	349.2	(61.1%)
YHT	Grassland	1073.9	-7.4	1066.6	-48.5	1018.1	-39.0	979.0	46.6	1025.7	54.5	1080.2	6.2	(0.6%)
	Shrubland	400.6	-31.4	369.2	3.2	372.4	15.9	388.2	-71.0	317.2	-101.5	215.7	-185.0	(-46.2%)
	Mixed-wood	60.8	2.4	63.2	1.3	64.5	0.2	64.6	16.8	81.4	-28.1	53.3	-7.5	(-12.3%)
	Coniferous	1011.9	17.2	1029.1	43.6	1072.7	25.8	1098.5	-3.8	1094.6	96.7	1191.4	179.5	(17.7%)

* Images to derive land cover change from 1992-2003 not available for SC and TY

Appendix VIa. Probability of transition of spatially referenced land cover type between 1952 and 2003 for the Ya Ha Tinda, Alberta, Canada.

	2003									
1952	Grassland	Shrubland	Mixed-wood	Coniferous	Logged	Pond	River	Building	Road/ Bare	Pasture
Grassland	40.7	15.2	2.3	38.5	0	0.3	2.1	0.002	0.4	0.5
Shrubland	47.1	30.2	2.9	18.6	0	0.15	0.55	0.02	0.3	0.14
Mixed-wood	24.1	10.1	33.5	30.7	0.04	0	0.3	0.05		0
Coniferous	2.4	1.1	0.7	92.0	2.1	0	0.6	0	1.2	0.002
Logged	0	0	0	0	0	0	0	0	0	0
Pond	36.6	0.06	0	0.003	0	63.2	0	0	0	0
River	19.0	4.0	0.9	40.0	0	0	31.7	0	4.4	0
Road/Bare	47.0	5.4	0.02	37.3	0	0	0	0	10.3	0
Pasture	36.0	0.5	6.3	0.5	0	0	0	0.3	3.6	53.9

Appendix VIb. Transition matrix showing change (%) in land cover type between 1952 and 2003 for the West Lakes, Alberta, Canada.

	2003						
1952	Grass-land	Shrub-land	Mixed-wood	Coniferous	Pond	River	Road/ Bare
Grassland	23.1	9.7	1.4	63.5	0.05	0.02	1.9
Shrubland	14.6	25.4	0.95	57.2	0.12	0	1.7
Mixed-wood	17.8	17.8	28.8	52.0	0	0	0
Coniferous	1.5	1.7	1.23	94.3	0.2	0.07	1.04
Pond	4.23	3.2	1.06	23.5	68.0	0	0
River	8.2	24.4	37.5	4.2	0	0	25.7
Road/Bare	11.4	14.8	0	22.3	0	13.3	38.1

*logged areas and buildings not present

Appendix VIc. Transition matrix showing change (%) in land cover type between 1952 and 2003 for the Tyrrell Flats, Alberta, Canada.

	2003				
1952	Grassland	Shrubland	Coniferous	River	Road/Bare
Grassland	47.9	1.8	33.6	0.4	13.8
Shrubland	3.5	68.8	19.4	3.8	4.5
Coniferous	1.6	0.8	93.4	0.8	3.3
River	2.2	0.1	20.0	72.7	5.1
Road/Bare	2.1	0	19.0	8.5	70.4

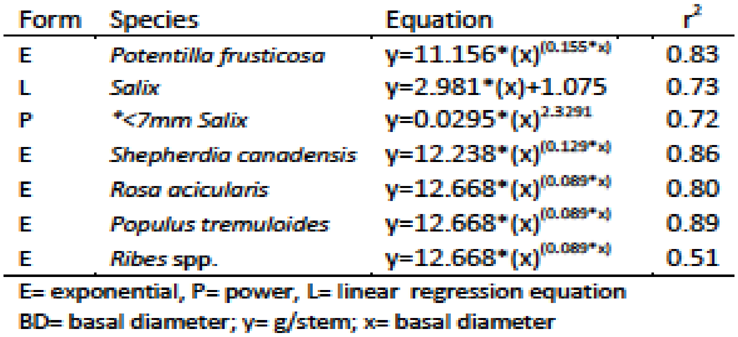
*logged areas, buildings, and mixed-wood not present, ponds exhibited 0 change

Appendix VId. Transition matrix showing change (%) in land cover type between 1952 and 2003 for Scotch Camp, Alberta, Canada.

	2003				
1952	Grassland	Shrubland	Coniferous	River	Road/Bare
Grassland	58.7	10.4	29.2	0.1	1.5
Shrubland	9.4	42.1	46.7	1.1	0.7
Coniferous	0.22	2.2	92.6	0.7	2.2
River	0	13.7	35.4	50.9	0
Road/Bare	0	0	0	0	0

*logged areas, buildings, mixed-wood and ponds were not present

Appendix VIII. Regression equations (Visscher et al. 2006) used to derive biomass for browse species within the Red Deer River Valley, Alberta, Canada. The equation for <7 mm *Salix* was derived from willow collected in the study area.

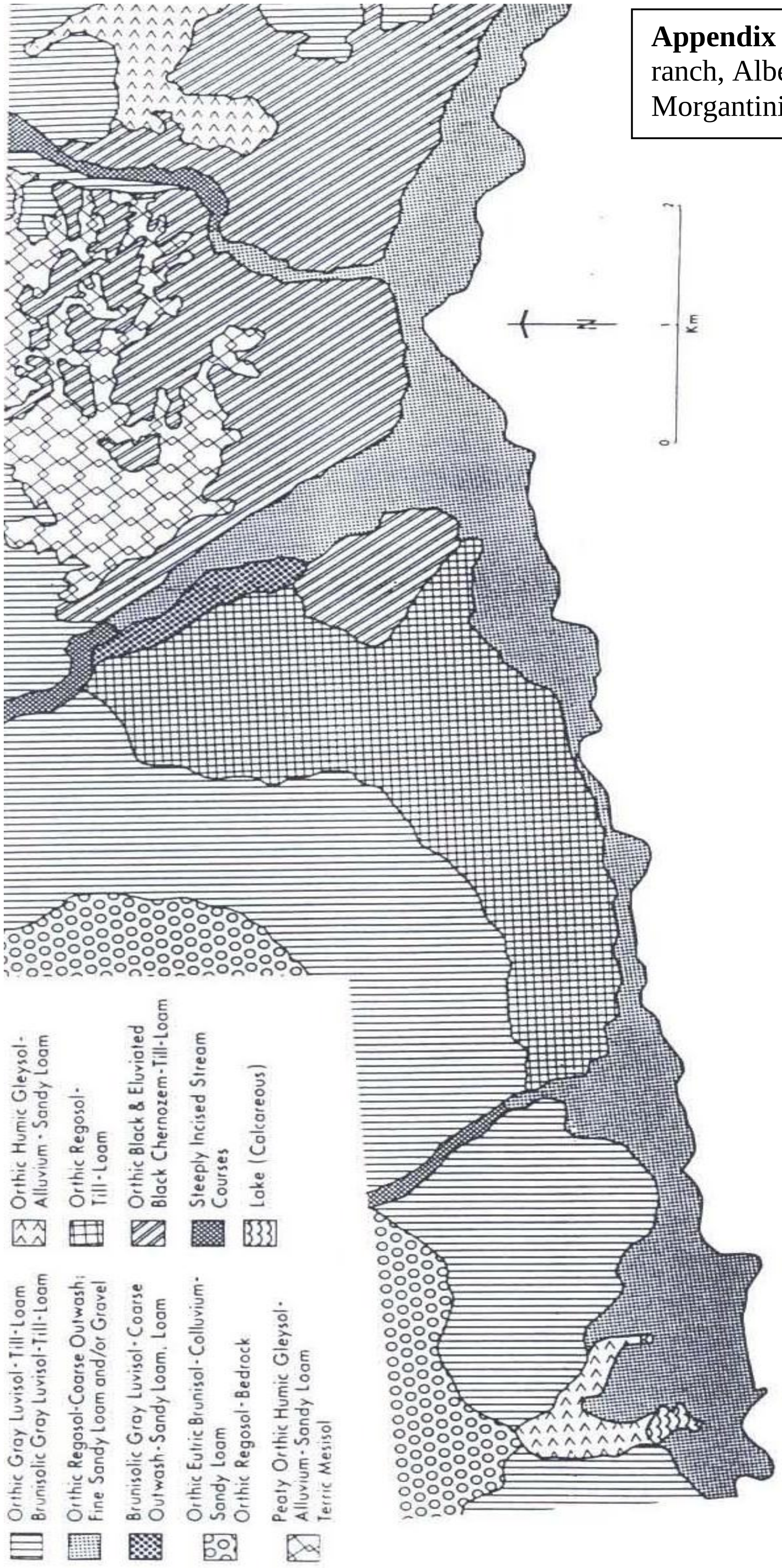


Form	Species	Equation	r^2
E	*Potentilla frusticosa*	$y=11.156*(x)^{(0.155*x)}$	0.83
L	*Salix*	$y=2.981*(x)+1.075$	0.73
P	**<7mm Salix*	$y=0.0295*(x)^{2.3291}$	0.72
E	*Shepherdia canadensis*	$y=12.238*(x)^{(0.129*x)}$	0.86
E	*Rosa acicularis*	$y=12.668*(x)^{(0.089*x)}$	0.80
E	*Populus tremuloides*	$y=12.668*(x)^{(0.089*x)}$	0.89
E	*Ribes* spp.	$y=12.668*(x)^{(0.089*x)}$	0.51

E= exponential, P= power, L= linear regression equation

BD= basal diameter; y= g/stem; x= basal diameter

Appendix VII. Soils on the Ya Ha Tinda ranch, Alberta, Canada, as presented in Morgantini 1995.



Soils and parent materials of the Ya-Ha-Tinda Ranch (after McGillis, 1976)

Appendix IX. All remaining radio-collared elk as of June 2011. Ear tag ID represents tag color: bl = blue, gr = green, or = orange, yl = yellow, and number. The letter G identifies elk with GPS collars.

EAR TAG	TAG	FREQ.	Beat	C - 1	C - 2	C - 3	Date Deployed	Comments
bl 204	R/L	150,710	Single	red	white	green	24-Feb-07	
bl 211	R	149,922	Double	green	blue	yellow	4-Feb-09	
bl 213	R/L	148,331	Double	red	green	black	27-Mar-05	
bl 220	R/L	152,060	Double	yellow	blue	red	24-Feb-08	
bl 221	R/L	148,786	Single	black	yellow	green	22-Mar-06	
bl 222	R/L	151.2409	single (40/min)	red	blue	yellow	23-Jan-11	
bl 226	R/L	149.586	Double	yellow	red	green	20-Mar-06	Mort signal near Windy
bl 234 G	R	150.6716	Single	black	white	red	19-Mar-10	
bl 236	None	151.919	single (40/min)	Black	red	white	23-Jan-11	No ear tags
bl 240	R/L	152,990	Single	yellow	purple	black	1-Mar-07	
bl 244	R/L	152,695	Double	yellow	yellow	yellow	1-Mar-07	
bl 245	R/L	148,532	Single	black	white	purple	1-Mar-07	
bl 246	R	148,641	Single	black	purple	green	1-Mar-07	
bl 251	R/L	150,754	Double	red	yellow	green	1-Mar-07	
bl 255 G	R/L	152,920	Single	white	red	black	19-Mar-10	
bl 260 G	R/L	152,889	Single	white	yellow	red	19-Mar-10	
bl 261	R/L	152,071	Single	blue	yellow	blue	24-Feb-08	
bl 262	R/L	148,312	Single	red	red	red	24-Feb-08	
bl 263	R/L	151,071	Single	yellow	red	black	24-Feb-08	Weak signal
bl265	R/L	150,850	Single	blue	white	green	4-Feb-09	
bl267	R/L	150,430	Single	red	green	yellow	4-Feb-09	
bl268	R/L	150,680	Single	black	black	white	4-Feb-09	
bl272	R/L	150,329	Single	green	red	red	4-Feb-09	

bl284	R/L	150,539	Single	yellow	red	blue	4-Feb-09	
bl285	R/L	150,439	Single	blue	white	yellow	4-Feb-09	
bl288	R/L	150,620	Single	green	yellow	green	4-Feb-09	
bl290	R/L	150,640	Single	blue	green	yellow	4-Feb-09	
bl292	R/L	150,690	Single	white	black	red	4-Feb-09	
bl295	R/L	150,729	Single	white	blue	red	4-Feb-09	
bl 622	R	152,790	Double	red	yellow	-	17-Feb-03	
gr 131	R/L	152,706	Double	green	green	-	17-Feb-03	Very Weak Signal
gr 133	L	148,970	Double	green	red	black	24-Feb-07	Very Weak Signal
gr 177	R/L	148,501	Double	-	-	-	20-Mar-06	No Signal
gr 183	L	152,750	Double	black	black	-	6-Apr-03	
yl 1	L	152,464	Double	red	-	-	17-Feb-03	
yl 2	R/L	148,980	Double	red	black	black	17-Feb-03	
yl 55	R/L	152,125	Double	black	black	black	6-Apr-03	Bow Valley
yl 62	R/L	149,206	Double	-	-	-	5-Feb-04	
or 1	R/L	150,500	single	red	red	black	19-Mar-09	
or 2	R/L	150,629	single	red	black	green	19-Mar-09	
or 3	R/L	150,470	single	black	red	blue	19-Mar-09	
or 5	R/L	150,669	single	red	white	black	19-Mar-09	
or 6	R/L	150,320	single	yellow	yellow	green	19-Mar-09	
or 7	R/L	150,560	single	black	white	yellow	19-Mar-09	
or 8	R/L	150,699	single	blue	white	black	19-Mar-09	
or 9 G	R/L	150,800	single	green	yellow	red	19-Mar-10	
or 10	R/L	150,868	single	red	green	yellow	19-Mar-09	
or 11	R/L	150,929	single	black	yellow	blue	19-Mar-09	
or 12	R/L	150,549	single	yellow	blue	white	19-Mar-09	
or 13	R/L	150,839	single	black	red	green	19-Mar-09	Missing
or 14	R/L	150,789	single	black	white	black	19-Mar-09	

or 15	R/L	150,858	single	white	blue	red	19-Mar-09
or 16	R/L	150.379	single (40/min)	red	yellow	blue	23-Jan-11
or 17	R/L	151.0802	single (40/min)	red	blue	white	23-Jan-11
or 18	R/L	151.061	single (40/min)	red	white	blue	23-Jan-11
or 19	R/L	151.2808	single (40/min)	yellow	red	yellow	23-Jan-11
or 20	L	151.1987	single (40/min)	red	yellow	red	23-Jan-11
or 21	R/L	150.6	single (40/min)	white	blue	black	23-Jan-11
or 22	R/L	150.3409	single (40/min)	yellow	blue	yellow	23-Jan-11
or 23	R/L	150.619	single (40/min)	green	yellow	white	23-Jan-11
or 24	R/L	151.26	single (40/min)	blue	red	blue	23-Jan-11
or 25	R/L	150.421	single (40/min)	black	red	yellow	23-Jan-11