UNIVERSITY OF CALGARY

Forest and Tree Encroachment into Fescue Grasslands on the Cypress Hills Interprovincial Park – Alberta Plateau

by

Kerri Jean Widenmaier

A MASTER'S DEGREE PROJECT SUBMITTED TO THE FACULTY OF ENVIRONMENTAL DESIGN IN PARTIAL FULFILLMENT OF THE REQUIREMENTS FOR THE DEGREE OF MASTER OF ENVIRONMENTAL DESIGN (ENVIRONMENTAL SCIENCE)

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FACULTY OF ENVIRONMENTAL DESIGN

The undersigned certify that they have read, and recommend to the Faculty of Environmental Design for acceptance, a Master's Degree Project entitled "*Forest and Tree Encroachment into Fescue Grasslands on the Cypress Hills Interprovincial Park – Alberta Plateau*" submitted by Kerri Jean Widenmaier in partial fulfilment of the requirements of the degree of Master of Environmental Design (Environmental Science).

Supervisor, Dr. Wayne L Strong,
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ABSTRACT

Forest and Tree Encroachment into Fescue Grasslands on the Cypress Hills Interprovincial Park – Alberta Plateau

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Submitted in partial fulfillment of the requirements of the degree of Master of Environmental Design in the Faculty of Environmental Design, The University of Calgary

Supervisor: Dr. Wayne Strong

September, 2009

Forest and tree encroachment into fescue grasslands is an ecological concern on the plateau of Cypress Hills Interprovincial Park, Alberta, Canada. A combination of field sampling (109 transects), and dendrochronological (n = 1,361 trees) and remotely-sensed image analysis (1950, 1971, 1981, 1992, and 2002) techniques was used to determine the extent and rate of forest and tree encroachment. Forest cover increased by 52% between 1950 and 2007, and this area of increase represented 11% of the plateau. The area of ingress (isolated trees and surrounding grassland) occupied an additional 9% of the plateau. *Pinus contorta* was the most common tree species associated with encroachment. Invasion occurred in a "nonlinear" pattern where encroachment increased in density by filling spaces around and between previously established trees. Tree establishment increased at an exponential rate since the last stand replacing fires of the late 1800s. Average annual temperature increased 0.6°C between 1929 and 2005, with winter and spring temperatures accounting for most of the increase, and annual total

precipitation increased by 85 mm. Establishment numbers of *Pinus contorta, Picea albertiana,* and *Populus tremuloides* appeared to be related to certain climatic variables, and the most important of these variables was spring temperatures for *Pinus contorta* establishment. A landscape devoid of wildfires, combined with increased moisture availability and a longer frost-free season, likely created an environment conducive to the establishment and development of coniferous trees and forests on the plateau. Conservation of the fescue grasslands should be possible by mimicking the long-term fire regime.

Key Words: climate change, Cypress Hills, fescue grassland, *Festuca campestris*, forest encroachment, fire suppression, grazing, *Pinus contorta*

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Thank you to Cameron Lockerbie of Sustainable Resource Development for overseeing this project from start to finish, and to Les Weekes of Cypress Hills Interprovincial Park – Alberta, who always promptly responded to my questions and requests. Lorna Allen (Alberta Natural Heritage Information Centre) facilitated the identification of this project through her professional contacts. My field work was possible due to the hard work of my field assistant and very good friend Joanna James. The kind assistance of everyone at Cypress Hills Inteprovincial Park, including Julie MacDougall and Randy Woolsey, made my stay at the Park an enjoyable and rewarding experience.

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CHAPTER 1: INTRODUCTION

Forest and tree encroachment into fescue grasslands is an ecological concern in the Cypress Hills of southeastern Alberta (Western Ecological Services Ltd. 1980; Korpela 2001; O2 Planning + Design Inc. 2007). Fescue grassland is considered an endangered ecosystem by Environment Canada (Trottier 2002), because <5% of its original extent remains (Holcroft Weerstra 2003). A combination of ecological processes and land-use activities such as fire suppression, plains bison (*Bison bison bison*)[1] extirpation, agricultural and urban development, overgrazing, exotic species invasion, and woody plant encroachment have either destroyed or irrevocably altered these ecosystems (Bailey and Wroe 1974; Campbell *et al.* 1994; Vujnovic 1998; Downing and Pettapiece 2006). Fescue grasslands are an important element of biodiversity in the Cypress Hills Interprovincial Park in Alberta (CHIP-AB). For this reason, an objective of a recently drafted management plan was to "conserve the extent, health, and native biodiversity of the fescue grasslands" (O2 Planning + Design Inc. 2007). Besides contributing to landscape diversity, fescue grassland provides important habitat for Sprague's pipit (*Anthus spagueii*), which is a species listed as "special concern" by Alberta's Endangered Species Conservation Committee (2003) and "threatened" by COSEWIC or the Committee on the Status of Endangered Wildlife in Canada (2009).

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[1] Plant and animal nomenclature follows the Integrated Taxonomic Information System (Canadian Biodiversity Information Facility 2009), except *Picea albertiana* which was based on recent research by Strong and Hills (2006).

The grasslands provide important habitat for mountain plover (*Charadius montanus*) and burrowing owl (*Athene cunicularia*), which have endangered status in Canada (COSEWIC 2009). Fescue grassland is a highly palatable and nutritious forage, supporting both wild and domesticated grazers (Bezeau and Johnston 1962). A substantive population of native ungulates reside in CHIP-AB (i.e., elk - *Cervus elaphus*, white-tailed deer - *Odocoileus virginianus*, and mule deer - *Odocoileus hemionus* – Hegel *et al.* 2009), which has permitted the recent colonization of the park by cougars (*Puma concolor*) (O2 Planning + Design Inc. 2007). The Park's grasslands have long been used by local stock associations as rangeland (Rangeland Conservation Service Ltd. 2001). Of the factors that can influence the long-term survival of fescue grasslands, forest encroachment is one of the least understood and more difficult to control from a resource management perspective. However, if fescue grassland continues to evolve to forest, biodiversity and important wildlife habitat could be permanently lost due to changes in vegetation and to soil properties that may make restoration very challenging (Amiotti *et al.* 2000; Lang and Halpern 2007).

Discussion and research regarding forest and tree encroachment in the Cypress Hills has occurred for decades. Newsome and Dix (1968) reported that *Populus tremuloides* (trembling aspen) appeared to be invading the grassland >40 years ago. Scace (1972) discussed the possible relationship between tree encroachment and fire suppression in a study of the history of use and management of the Cypress Hills. The early 1990s brought renewed interest; the issue was included in a study by Western Ecological Services Ltd. (1991) and was a topic of discussion at the Cypress Hills Forest

Management Workshop held in Medicine Hat, Alberta (Peterson and Peterson 1992). Both the study and workshop identified the need for research and improved documentation of the ecological processes associated with tree establishment in fescue grasslands. Hull (2002) reiterated the need for forest encroachment research, and its relation to grassland bird habitat selection, species diversity, and population dynamics over time. Field and spatial analysis studies have been attempted to quantify forest encroachment (Korpela 2001), but this research was not adequate to guide park management decisions. This study only evaluated forest islands in one area of the park where encroachment appeared to be concentrated, and the spatial analysis relied upon Alberta Vegetation Inventory (AVI) data. The AVI data was derived from 1:20,000-scale air photos, with an interpretation accuracy of 80% and no field auditing (Alberta Sustainable Resource Development 2008). In addition, AVI does not adequately represent savannah-like vegetation (Korpela 2001), nor did it identify the presence of tree seedlings or saplings. Additional research related to forest encroachment was identified as critical for resource management in the recently drafted Park management plan (O2 Planning + Design Inc. 2007).

Determining management options and strategies acceptable to the Park and the public will be a challenging endeavour. The general consensus is that fire suppression has permitted trees to encroach upon the grassland (Strauss 2001; O2 Planning + Design Inc. 2007), as has been surmised in other studies of forest encroachment (Burkhardt and Tisdale 1976; Briggs *et al.* 2002). If fire cessation is responsible for forest and tree invasion in CHIP-AB, a prescribed burning program could be an effective tool to lessen fescue grassland loss (Ducherer *et al.* 2009). However, the use of fire is controversial due to the presence of the village of Elkwater in the Park. Other management options, such as logging, may also not be looked upon favourably by the public (Locke 1992). Several studies have investigated forest encroachment in western Canada and other parts of the world (Wearne and Morgan 2001; Kennedy and Sousa 2006; Zier and Baker 2006; Coop and Givnish 2007). A review of this research provides little insight as to precedents regarding vegetation management options, because most studies focused on quantifying the amount of encroachment and linking it to causal factors.

The purpose of this research was to evaluate forest and tree encroachment into fescue grasslands on the plateau in Cypress Hills Interprovincial Park, Alberta, and to provide objective scientific information necessary to support informed decisions regarding vegetation management. This research attempted to answer the following four questions:

a) *What is the quantity and rate of fescue grassland loss on the plateau due to forest and tree encroachment during the 1950–2007 time period;*
b) *What patterns of tree establishment are associated with the encroachment;*
c) *What is the likely ecological or land-use mechanism that has promoted encroachment*; and
d) *What management practices could mitigate the loss of fescue grasslands?*

The first question considered whether trees were invading the fescue grasslands on the plateau, quantified the amount of historic invasion, and determined the rate of encroachment. Establishment patterns investigated included those associated with year of initiation, tree species, site conditions, and spatial distribution. Ecological processes and land-use mechanisms considered as potential causal factors for encroachment were livestock grazing, elk grazing, fire regime, and climatic trends. The results of the first three questions were used to identify potential management options to reduce or reverse the rate of tree and forest encroachment on the Cypress Hills Interprovincial Park – Alberta plateau.

This Masters Degree Project is intended as a reference for resource managers of Cypress Hills Interprovincial Park – Alberta when making decisions regarding vegetation management on the plateau. The following three chapters outline the approach taken to investigate forest encroachment (Chapter 2), and present (Chapter 3) and discuss (Chapter 4) the findings of the research. The final chapter (5) uses these findings to develop and consider alternative management options for mitigating fescue grassland loss due to forest encroachment.

CHAPTER 2: METHODS

Study area

The Cypress Hills are a topographic upland that is slightly inclined to the northwest and extends from southeastern Alberta into southwestern Saskatchewan. They cover an area of roughly 2,590 km^2, extending 132 km in length along an east-west axis, but never exceeding 24 km in width. The highest elevation (1,468 m) occurs in Alberta on the west end of the upland, 360 m above the surrounding plains, and is the highest elevation in Canada between the Rocky Mountains and the Labrador Highlands. The Cypress Hills form part of the continental divide between two major North American drainage systems: the Saskatchewan River, which flows north to Hudson Bay; and the Mississippi River, which drains southward to the Gulf of Mexico.

A large portion of the Cypress Hills is an interprovincial park jointly managed by the provinces of Alberta and Saskatchewan (O2 Planning + Design Inc. 2007) (Figure 1). The Alberta portion of Cypress Hills Interprovincial Park (CHIP-AB) is located approximately 45 km southeast of Medicine Hat, Alberta, Canada, and consists of side-slopes and a plateau. The study area occurred entirely within CHIP-AB, and roughly coincides with the boundaries of the plateau, which represented approximately 78 km^2 (Figure 1). The study area boundaries were approximately 49°35–40'N, 110°0–23'W and elevations ranged from 1,334–1,468 m, with an average elevation of 1,415 m.

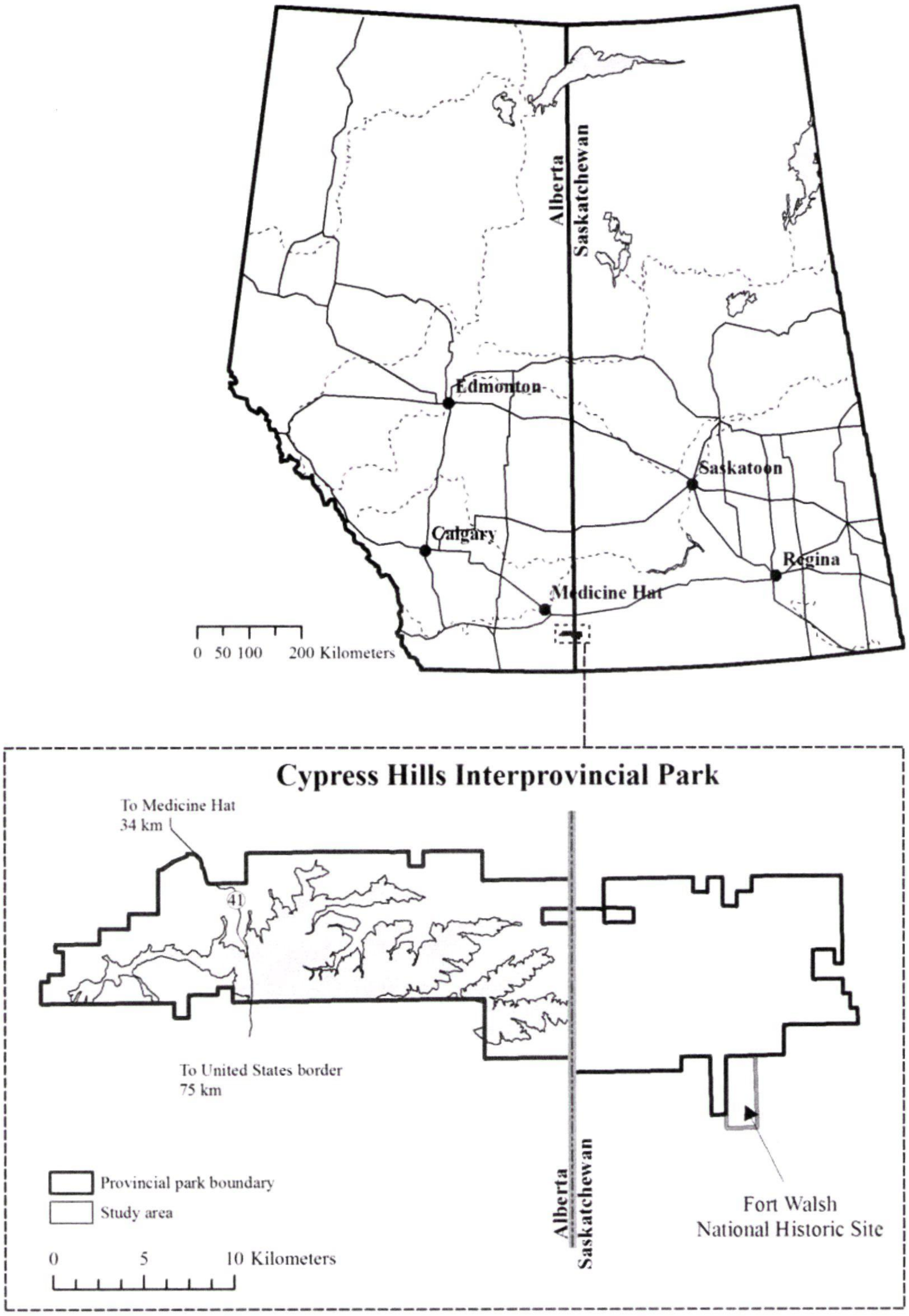

Figure 1. Location of study area within the western portion of Cypress Hills Interprovincial Park.

The Cypress Hills are a remnant of a Tertiary alluvial plain that, due to topographic inversion, now extends above the surrounding plains (Westgate 1968; Catto 1981; Vreeken 1986, 1990; Vreeken and Westgate 1992). The western end of the plateau was a nunatak ~125 m above the adjacent Wisconsinan ice sheet during the Pleistocene (Dyke and Prest 1987). Bold escarpments define the northern, western, and eastern limits of the plateau, whereas the southern limit slopes gradually downward to the surrounding plains. A system of short, steep streams arranged in a radial pattern drain runoff from the plateau into Battle Creek, which was once a glacial meltwater channel (Christiansen 1979). Several small, dry valleys formed by runoff occur on the plateau surface and southern slope (Newsome and Dix 1968). Intermittent streams with alkaline lakes flow north towards the South Saskatchewan River, although little water actually reaches the river, whereas substantive flow continuously feeds the Milk River drainage system to the south (Newsome and Dix 1968). A layer of Holocene loess up to 3.3 m thick has accumulated on the plateau since glacial times (Catto 1983), with mass wasting, surface erosion, and alluvial fan development further shaping the side-slopes (Vreeken 1990).

The Cypress Hills belong to the Montane Ecoregion of Alberta, which represents <1% of the province, mostly located in the Rocky Mountains (Strong 1992). This is in contrast to the surrounding plains, which belong to the Dry Mixed Grass Ecoregion. Temperatures typical of the Montane Ecoregion are warmest in July, with a summer monthly average of 11.9°C. Freezing temperatures can occur in all months, with the coldest month occurring in December (-7.9°C). Precipitation generally peaks twice; May to June and August to September (Strong 1992), with a yearly average of 589 mm (Downing and Pettapiece 2006).

Newsome and Dix (1968) estimated 20% of the Cypress Hills was forested in 1962–1964, but the remainder consisted predominantly of grasslands. The western and northern escarpments of the Cypress Hills were heavily forested, whereas the southern slope and summit of the plateau were primarily grassland intermixed with forest patches. Forests were concentrated in the northwest and dissipated southeastward. Breitung (1954) identified six major vegetation types in the area, with three found on the plateau: *i*) *Pinus contorta* (lodgepole pine) and *ii*) *Populus tremuloides* (trembling aspen) forests; and *iii*) *Festuca campestris* (rough fescue) grasslands. *Pinus contorta* was the most common tree on the plateau. Pure stands of *Populus tremuloides* sometimes occurred along the edge of *Pinus contorta* forest, and development of these trees was often stunted due to wind exposure (Breitung 1954). *Picea albertiana* (western white spruce) do not occur in pure stands in the study area, but some individuals were present in grasslands and *Pinus contorta* and *Populus tremuloides* forests (Breitung 1954; Newsome and Dix 1968). The type of grassland found above 1,230 m (Coupland and Brayshaw 1953) was not dominated by dry mixed grasses (*Stipa–Bouteloua* complex) of the surrounding prairies, but rather *Festuca campestris* grassland (Breitung 1954).

Grasses occurring with *Festuca campestris* in the study area included *Danthonia intermedia* (timber oat grass), *Elymus trachycaulus* ssp. *subsecundus* (slender wheat grass), *Festuca idahoensis* (Idaho fescue), *Helictotrichon hookeri* (Hooker's oat grass), and *Koeleria macrantha* (June grass). *Dasiphora floribunda* (shrubby cinquefoil) was abundant, and occurred in conjunction with *Lupinus argenteus* (silvery lupine) (Newsome and Dix 1968; Lewis 1980). Reference plant communities for the study area included the *Dasiphora floribunda/Festuca campestris – Danthonia intermedia* and *Dasiphora floribunda/Festuca campestris – Festuca idahoensis* types (Willoughby *et al.* 2005). Loamy Orthic Black Chernozems[2] underlaid *Festuca campestris* vegetation, whereas Dark Gray Luvisols occurred with *Pinus contorta* and *Populus tremuloides* forests (Greenlee 1981).

Fire once shaped the vegetation distribution and composition of the Cypress Hills. The fire frequency for the Cypress Hills during the time period 1740 to 2000 was approximately 45 years (range 24 – 65), with frequent, low intensity burns, and infrequent, large intensity fires (Strauss 2001). Most current forest in the Cypress Hills established following stand replacing fires that occurred in the 1880s (Strauss 2001). Newsome and Dix (1968) marked the year 1889 as "the cessation of fire as an active ecological force in the Cypress Hills", and by 1911 all wildfires were suppressed in the Park (Scace 1972). Since suppression began, fires have been extinguished with little burning of trees, with only a couple of exceptions (Scace 1972). During the period 1969 to 2006, 63 wildfires occurred within the Park (Alberta Sustainable Resource Development 2006) and only one fire affected the study area; 35–40 ha of fescue grassland burned in 1992 (L. Weekes, Alberta Sustainable Resource Development, 24 June 2009, pers. comm.). Of these wildfires, 93% were human-caused, most due to

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[2] Soil nomenclature is consistent with the current Canadian system of soil classification (Soil Classification Working Group 1998).

recreation, with park management activities being the second leading cause of wildfires. Prior to suppression activities, humans may have been responsible for most of the wildfires that occurred in the area (Oetelaar and Oetelaar 2008).

Before the arrival of Euro-canadians in the late 1800s, the Cypress Hills were occupied and managed by indigenous Plains people (Scace 1972). These people strongly influenced the ecological development of the area; they frequently ignited fires to maintain grassland forage for horses and bison, and may have been partially responsible for the introduction of montane vegetation (Oetelaar and Oetelaar 2008). European settlement began in the 1870s. The Alberta portion of the Cypress Hills was managed as a 200 km^2 Dominion Forestry Preserve by Alberta Lands and Forests after 1911 (Scace 1972), and became part of Canada's first interprovincial park in 1989. Agriculture is the dominant land-use affecting CHIP-AB. Livestock grazing has occurred on the grasslands since the 1890s (Western Ecological Services Ltd. 1980). Currently, three stock associations, all established in 1919, graze cattle within the park (Western Ecological Services Ltd. 1980; Rangeland Conservation Service Ltd. 2001).

In addition to cattle, elk are an important grazer within the Park. Elk were extirpated from the Cypress Hills by the early 1900s (Soper 1946), and reintroduced in 1938 (Keith 1977). The population grew and expanded throughout the Cypress Hills following reintroduction, and ensuing conflicts necessitated the implementation of annual management hunts in both provinces (Hegel 2004). These hunts were aimed at maintaining a population of 700 individuals, and continue to the present day (Hegel

2004). Since the implementation of the hunt, the elk population has fluctuated between ~600 and ~930 individuals.

Site selection and sampling design

Site selection

Prior to field work, the oldest available aerial photographs of the study area, taken at an acceptable scale for vegetation interpretation, were used to locate 200 potential sampling transects. This allowed the study of forest and isolated tree development over the longest possible time period. For the purposes of this study, forest was defined as the formation or expansion of relatively contiguous stands of trees with touching crowns, whereas isolated trees represented the ingress of individuals into nonforest vegetation. Photos taken in 1950 at a scale of 1:40,000 (Table 1) were scanned at a resolution of 1,200 dots per inch (dpi) as a basis for locating transects using the Environmental Systems Research Institute program ArcGIS 9.2 (ESRI Ltd. 2006). ArcGIS, a computer-based geographical information systems program, was used for spatial analysis, data management, and mapping.

Transects were digitized onto the 1950 air photos. Most transects were located along the perimeter of the plateau and extended from the 1950 forest boundary into the grassland-dominated plateau surface. Where necessary, transects included a portion of the side-slope in order to reach the 1950 forest boundary. In these cases, transects were located based on uniformity of the side-slope and perpendicular to the contour. Each transect was 200 m in length, unless it crossed the entire width of the plateau. To ensure

Table 1. Air photos and remotely-sensed images used for spatial analysis of the Cypress Hills Interprovincial Park – Alberta plateau.

Year	Scale/ Resolution	Flight line or Image type	Line	Photos
1950	1:40,000	4911-2194A	-	106-113
		4912-2194A	-	52, 54
		4911A-2191	-	64
1971	1:12,000	AS1103	1	33, 35, 37
			2	72, 74, 76, 78, 80, 82
			3	99, 101, 103, 105, 109, 111, 113, 115, 117, 119, 121, 123, 125
			4	140, 144, 146, 148, 150, 152, 154, 156
			5	171, 173, 175, 177, 179, 181, 183, 185, 187, 189, 195
			6	218, 220, 222, 226, 228, 230
			7	265, 267, 269, 271
			8	305
1981	1:10,000	AS2343	3	96, 98, 100
			4	112, 114, 116, 118, 120, 124, 126, 128, 130, 132, 134, 136, 138, 140, 142, 144
			5	153, 155, 157, 160, 164, 166, 170, 172, 174, 178, 180
			6	197, 199, 201, 203, 205, 207, 209, 211, 213, 215
			7	236, 238, 240, 242, 244
			8	14, 16, 20
1992	1:15,000	AS4312	1	41, 43
			2	65, 67, 69
			3	78, 81, 83, 85, 87, 89, 91, 92, 95, 97, 99, 101
			4	113, 115, 117, 119, 121, 123, 125, 127, 138, 140, 142, 144, 146
			5	138, 140, 142, 144, 146
2002	0.5-m resolution	Green Vegetation Index	-	401007, 401008, 402008, 403008

that no transect crossed more than one 1950 forest boundary, any transect that crossed the width of the plateau was subdivided mid-way through the grass-dominated section. The location of the plateau surface and side-slopes were considered during transect placement using contour lines overlaid on the 1950 photos from a *contournad83* shapefile layer, provided by Alberta Parks and Recreation.

Transects covering forest islands and grassland vegetation were also sampled. These transects did not begin at the 1950 forest boundary, although sometimes included pre-1950 trees. They were located on the most recent remotely-sensed images: 0.5-m Green Vegetation Index images taken in 2002, which were provided by Alberta Parks and Recreation. Forest island transects began in grassland, bisected forest patches, and ended in grassland. Grassland transects began and ended in grassland, and were not intended to include forest patches. Both of these latter types were located solely on the plateau surface, and comprised 10% of the pool of potential transects. From the pool of potential transects, 100 were selected for sampling using a random number generator.

Field sampling

Coordinates for the first sampling point of each transect were uploaded into a Garmin Geko 201 Global Positioning System (GPS), which was used to locate them in the field. The first sampling point along each transect was located on the 1950 forest boundary, unless the transect covered a forest island or grassland vegetation. Each transect then extended in a straight-line in a predetermined direction and was located using a compass. For ground truthing purposes, a new coordinate was recorded on the GPS at the first and last sampling points in the field.

Sampling occurred at 10-m intervals along one side of each transect. The side with the most level topography was sampled, and if both were level, one side was randomly chosen by flipping a coin. At each sampling point, three size-classes of trees[3] based on basal diameter were measured: mature ($\geq$50 mm), saplings (25-49 mm), and seedlings (<25 mm). Only the nearest representative of each size-class located at a right-angle ($\pm$50 cm) and within 15 m of the transect was sampled. For each tree, its distance from the transect, species, basal diameter, and height were determined. Basal diameter was measured to the nearest millimeter using a ruler, whereas height was measured to the nearest centimeter using a measuring tape or to the nearest decimeter with a clinometer when trees exceeded 2 m in height. Increment cores to determine age were collected from mature trees and stored in plastic straws, whereas basal disks were collected from saplings and seedlings and stored in paper envelopes. All trees were sampled immediately above the root collar to maximize the accuracy of age determinations.

At the first and every subsequent third sampling point, the topographic slope gradient and aspect were measured, and percent shrub and herbaceous plant cover were estimated, with the dominant shrub identified in a 2-m $\times$ 2-m quadrat. Soil Ah-horizon thickness was measured in the lower-right corner of each quadrat using a 2.5 cm diameter corer. Quadrat and tree sampling occurred on the same side of the transect.

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[3] Tree-forming species (*Picea albertiana, Pinus contorta, and Populus tremuloides*) of any size are collectively referred to as trees.

Transects were 200 m long unless: (*i*) the width of the plateau was less than this length or (*ii*) a tree was encountered within the last 60 m of the transect. When the first situation occurred, the transect was subdivided as previously described. If the second occurred, an additional 60 m was added to the transect length until no additional trees were encountered. The number of sampling points increased with transect length.

Tree aging

Tree increment cores and basal disks were processed using standard dendrochronological techniques (Stokes and Smiley 1968). To facilitate the counting of annual growth rings, cores were glued to wooden mounts and sanded, first with a belt sander, and then a block of wood. Clipped stems were only block-sanded. Progressively finer sand paper was used in both cases to improve the visibility of tree growth rings. Aging was conducted using a binocular dissecting scope. When cores did not intersect the pith, missing rings were estimated using species-specific templates derived from complete cores with similar ring patterns (Haugo and Halpern 2007). These templates were drawn on trace paper and overlaid on the core. No more than four rings were added to the count using this method. The age of partially rotten or incomplete cores were estimated from field measured basal diameters, based on species-specific regression models developed from the pool of aged samples (Haugo and Halpern 2007). The regression equations and models used to estimate tree ages are presented in Appendix A.

Temporal and spatial analysis

Image analysis

Temporal and spatial analyses of air photos and satellite images were conducted to determine the rate and extent of encroachment between 1950 and 2002. For the purposes of this study, forest developed or trees established after 1950 are referred to as "new forest" and "new trees", respectively. The term "encroachment" was used to encompass all new forest, isolated trees, and areas of ingress. The latter represented areas of low density juvenile trees of various heights in non-forest vegetation. Maps depicting forest cover were created for 1950, 1971, 1981, 1992, and 2002 based on air photo and satellite coverage. Table 1 provides details regarding the images used. These years were chosen for analysis because the image sets were taken at similar time intervals at an adequate resolution for vegetation analysis. A set of images for the 1960s would have been useful for this analysis, but were unavailable.

To allow the comparison of different image sets and other spatial data, individual images were georeferenced to the Universal Transverse Mercator grid based on the 1983 North American datum of the Green Vegetation Index images. Features common to more than one set of images were used as reference points for registering each photo. From 8 to 15 identifiable features such as road intersections, buildings, and prominent permanent landscape features such as rock outcrops were used as control points.

After georeferencing, raster images depicting encroachment were created. The first step in creating these images was to identify a spectral signature that represented forests and trees on each photo. The photos were then reclassified from their original grey-scale values, using the ArcGIS *Reclassify* tool in the Spatial Analyst extension, to create a binary classification of treed and non-treed areas. Each individual photo was reclassified using a different spectral signature to avoid errors in vegetation classification, which resulted from different levels of contrast among photos. In some cases, topography necessitated the reclassification of portions of a photograph due to different levels of contrast created by aspect differences. Spectral signatures were identified by choosing a numerical grey-scale threshold that best represented encroachment for each individual photo or portion of a photo. This involved comparing the binary classification with the original photographic image, and continually adjusting the grey scale cut-off point until the boundaries of the binary image closely matched those of the forest boundary.

The forest cover images were joined using the *Mosaic* tool in ArcGIS to produce a single raster image for the entire study area for a given year. Each raster image was converted to a polygon layer using the *Conversion* tool. The resultant polygons were scrutinized for errors, and where necessary, incorrect delineated forest and tree cover polygons were removed or replaced with manually digitized polygons that more accurately depicted the extent of encroachment on the original image.

To segregate isolated trees from forests on the plateau, the *Aggregate Polygons* tool was used to group polygons within 10 m of each other that, when aggregated, resulted in a polygon of 100 m^2 or more. This aggregation distance was chosen because at 30% tree coverage the vegetation structure is considered to be woodland (Mueller-Dombois and Ellenberg 1974, p. 473). Aggregated polygons represented forest, whereas all others represented isolated trees. The *Calculate Geometry* tool was then used to determine the total area of forest and isolated trees for a given year.

Ground truthing

To ensure the most current year of spatial analysis accurately reflected forest cover on the ground, it was compared against data collected in the field. Coordinates collected in the field were used to digitize the transect locations on the 2002 forest cover map, including each 10-m sample point and perpendicular sample lines extending to 15 m. Each tree in the $\geq$50 mm size-class encountered during field sampling was located on these digitized transects, and it was determined whether or not it was represented by a forest or isolated tree polygon on the forest cover map. Only trees in this size-class were used, because trees in the smaller size-classes may not have been discernable within grasslands on the air photos. Each sample point after the last tree encountered on each transect, therefore located in grassland vegetation, was also located on the digitized transects to ensure that it was not represented by a forest or an isolated tree polygon.

Encroachment area calculation

An areal estimate of encroachment, or new forest plus the area of tree ingress, on the plateau from 1950 to 2007 was calculated from the transect data by determining the distance between the pre-1950 tree line and the last tree encountered on each transect. Area was determined by multiplying the average amount of encroachment encountered among all transects by the length of the 1950 tree line (Equation 1). The 1950 tree line was determined based on the 1950 forest cover map, and was also represented by trees older than 57 years from the transect data.

Equation 1: Area (ha) = $(L \times W) * P/10{,}000$

L = pre-1950 forest perimeter (m), which was 192.3 km

W = average width of invasion (m)

$P = t/N$

t = number of transects with a given characteristic

N = Total number of sampled transects

The perimeter length was determined using the *Calculate Geometry* tool in ArcGIS. The area of tree ingress was considered equal to the difference between the total encroachment minus the area of new forest.

Stand density calculation

For this research, tree stands were defined as three or more trees >50 mm in basal diameter encountered consecutively in the area of encroachment. Stand density or the number of trees per hectare was determined using Equation 2:

Equation 2: Stand density (trees/ha) = $\left[\ 100\ /\ (\sum d_j\ /\ \sum n_j)\ /\ 100\ \right]^2$

d = distance to the specimen (cm) from the transect line. If no specimen occurred at a sampling point, a value of 1500 cm was used.

j = basal diameter size-class

n = number of sample points in a stand

This method of density calculation was in part based on the point-centered quarter method (Mueller-Dombois and Ellenberg 1974, p. 110).

Statistical analysis

To determine the appropriate statistical approach, field data were tested for normality based on skewness and kurtosis. Data falling outside the ± 0.9 and -0.4 to +1.8 ranges, respectively (Wetherill 1981, p. 9), was considered to not have a normal distribution, which was typical of the data. Therefore, Kruskal-Wallis tests were used to identify significant differences among tree ages by species and among site characteristics. Probability values of ≤ 0.050 were considered statistically significant. Scheffé rank tests (Miller 1966, p. 166 - formula 110) were used to distinguish which groups differed within significant Kruskal-Wallis tests:

Equation 3: $|R_i - R_{i'}| < \sqrt{\chi^2} * \sqrt{(N(N+1))/12} * \sqrt{1/n_{i1} + 1/n_{i2}}$

Where:

R = mean rank of a group derived from Kruskal-Wallis test

χ^2 = chi-square for number of groups – 1 (df) based on α 0.050 level

N = total number of samples ($n_1 + n_2 \ldots n_i$)

n = number of samples in a group

i_1 = group 1 of comparison pair

i_2 = group 2 of comparison pair

The Statistical Package for Social Sciences (SPSS Inc. 2007) was used to compare grouped data, except for Scheffé rank tests, which were manually conducted.

Linear and nonlinear regressions, in combination with scatter diagrams and Pearson's product-moment coefficient of correlation (r or R), were used to summarize the degree of fit between independent and dependent variables, identify predictor variables of tree establishment numbers; and to model the relationship between these variables and tree establishment numbers. The coefficient of determination (r^2 or R^2) was used to indicate the amount of explained variance association with correlations and standard error of the estimate (SEE) was used to indicate the amount of error associated with regression models.

Site characteristic analysis

Tested site characteristics were percent herbaceous and shrub cover, shrub type, and Ah-horizon thickness based on grassland, pre-1950 forest, and encroachment vegetation types. For these analyses, new forest and areas of ingress were combined, i.e., encroachment, due to the difficulty in identifying a definite boundary between the two types, and the low number of sites in the area of tree ingress. Pre-1950 forest was differentiated from new forest based on the age of sampled trees. If the transect extended from the 1950 tree line, and within the first 30 m a tree older than 57 years was sampled, all sites prior to this tree were classified as pre-1950 forest. Subsequent sites were considered pre-1950 forest until no more trees older than 57 years were encountered, or more than 30 m occurred before the next tree of this age. If, within new forest, three or more consecutive sites had trees >57 years old, they were classified as pre-1950 forest. Otherwise, these older trees were considered part of the new forest.

Causal factor analysis

Climatic data compilation

Climate data were obtained from Environment Canada (2008). No single meteorological station within CHIP provided a continuous record for the years 1927 to 2007. This timeframe was considered long enough to identify climate trends that may have occurred prior to 1950-2007 for comparison with tree establishment numbers. Climatic trends were estimated for the Cypress Hills plateau from the nearest station with the longest and most continuous record, which was the Medicine Hat Airport (Station

3034480). Two stations, located 40 km from the study area and 175 m in elevation from each other, when combined provided the longest possible meteorological record for the Cypress Hills area. Station 4032000 (Cypress Hills Park) occurred at an elevation of 1,371 m and provided a discontinuous record from 1918 to 1972, whereas station 4021996 (Cypress Hills) occurred at an elevation of 1,196 m and had a discontinuous record from 1981 to 2007.

The two Cypress Hills stations in combination with Medicine Hat data were used to estimate temperature and precipitation values for the study area. A single missing temperature record in the Medicine Hat Airport data was determined by averaging the values from the same month of the preceding and following year. Also, problems with the precipitation data from June 2006 to December 2007 for Medicine Hat Airport station necessitated the direct use of Medicine Hat RCS data (Station 3034485), which was located 1 km from the airport. No attempt was made to infill missing data within the two Cypress Hills meteorological records.

The estimation of temperatures for the plateau involved three steps. The first step was to adjust Station 4021996 records to approximate those of Station 4032000, because the latter was closer in elevation to the study area. This adjustment was accomplished by separately regressing both Cypress Hills station records against the Medicine Hat record to obtain two linear models (Appendix B, Figures B1 and B2), and determining the difference between their y-intercepts (0.716°C). Combined Station 4032000 and adjusted Station 4021996 provided a dataset for the 1918 to 2007 time period in Cypress Hills Interprovincial Park.

The second step was to modify the combined set of Cypress Hills records to account for the difference in elevation between station 4032000 (~1,371 m) and the mean elevation (~1,415 m) in the study area. This was done by subtracting 0.434°C from all Cypress Hills temperature values, assuming an adiabatic rate of 1°C per 100 m rise in elevation (Strahler and Archibold 2008). The final step involved regressing the resultant set of data against Medicine Hat Airport record to obtain a linear equation for estimating temperature within the study area. The two sets of temperature records were an excellent fit ($r = 0.992$, $P < 0.001$, $n = 626$; Figure B5). Precipitation records for the study area were estimated using the same method, but no modification was made to account for the difference in elevation, because no model for this adjustment was available, and the precipitation trends for both Cypress Hills stations were similar. The fit of the two sets of precipitation records was less optimal than the temperature data, but still acceptable ($r = 0.754$, $P < 0.001$, $n = 627$; Figure B6). Additional scatter diagrams and regression models that illustrate the fit between the Cypress Hills and Medicine Hat data are presented in Appendix B.

The climate variables compiled for the study area included mean annual temperature, mean seasonal temperatures, total annual precipitation (P), total seasonal precipitation, potential evaporation (PE), and net water balance. Net water balance was calculated as P – PE using day-length adjusted potential evaporation index values based on Thornthwaite and Mather (1957) and Dunne and Leopold (1978):

Equation 4: $PE_{adjusted} = \sum 1.6 \text{ x } 10 \, [10T/I]^b \text{ x } d$

$j = 1$

where PE is potential evaporation per month (mm/month) adjusted for day-length; and T is mean monthly air temperature (°C).

$$I = \text{Annual Heat Index} = \sum_{j=1}^{12} [T / 5]^{1.514}$$

if $T \leq 0°C$ then $I = 0$

where j is the monthly value from November to October;

$b = 0.49 + 1.79 \text{ x } 10^{-2}(I) - 7.71 \text{ x } 10^{-5}(I^2) + 6.75 \text{ x } 10^{-7}(I^3)$; and

d is the day-length adjustment of radiation received during the middle of each month (Dunne and Leopold 1978, p.138)

Although more rigorous evapotranspiration models have been developed, such as the Penman-Monteith equation, they required additional meteorological and vegetation details such as net radiation, wind speed, and canopy conductance (Monteith 1965), which were unavailable for this research. P – PE values were standardized (i.e., x_i – mean/standard deviation) to set the mean net water balance equal to zero, and allow recognition of deviations from the normal condition. All annual climatic values were summarized for November to October, as November was the first month of the year after the growing season with a mean temperature below 0°C. To facilitate the identification and illustration of trends, five-year running averages were used based on averaging the previous two and following two yearly values. This approach meant that two years were lost at each end of the data sets.

Comparison of tree establishment with climatic trends

Tree establishment numbers were compared against climatic trends for the 1950 to 2001 time period. This time period was chosen because it coincided with the air photo analysis data, which allowed comparison of forest and isolated tree cover change with climatic trends and tree establishment numbers. Because numbers of seedlings of all species may not necessarily reflect the number of long-term surviving trees, trees under five years of age were excluded from the analysis, as was the approach taken by Hessl and Baker (1997). Several different climatic variables were developed to characterize changes in annual and seasonal precipitation, temperature, and net water balance (Table 2). Different versions of these variables were developed because the influence of climate on tree establishment can sometimes involve a time-lag of one or two years (Bigler *et al.* 2007). Nonmetric multidimensional scaling (MDS) ordination using STATISTICA software (Statsoft 1995) was used to identify the climate variables from Table 2 that were most closely associated with tree establishment by species, based on ordination distance (i.e., variables located nearer to the species establishment data based on Euclidean ordination distance were more closely related). Explained variance for the MDS ordination (Schiffman *et al.* 1981, p. 92) was calculated as:

Equation 5: $\text{explained variance} = 1 - (\text{total stress})^2$

Table 2. Climate variables compiled for comparison with tree establishment data.

Version of Variable	Variable description Tree establishment value coincides with climate variable value:	Climate variable names*		
		Precipitation	Temperature	Net water balance
Raw data	of same year	ppt_anu ppt_spr ppt_sum ppt_fal ppt_win	tem_anu tem_spr tem_sum tem_fal tem_win	P-PE_anu P-PE_spr P-PE_sum P-PE_fal
5 year running average	of same year averaged with preceding and following 2 years	ppt_anu_5 ppt_spr_5 ppt_sum_5 ppt_fal_5 ppt_win_5	tem_anu_5 tem_spr_5 tem_sum_5 tem_fal_5 tem_win_5	P-PE_anu_5 P-PE_spr_5 P-PE_sum_5 P-PE_fal_5
Lagged 1 year	of previous year	ppt_anu_dis1 ppt_spr_dis1 ppt_sum_dis1 ppt_fal_dis1 ppt_win_dis1	tem_anu_dis1 tem_spr_dis1 tem_sum_dis1 tem_fal_dis1 tem_win_dis1	-
Lagged 2 years	of two years previous	ppt_anu_dis2 ppt_spr_dis2 ppt_sum_dis2 ppt_fal_dis2 ppt_win_dis2	tem_anu_dis2 tem_spr_dis2 tem_sum_dis2 tem_fal_dis2 tem_win_dis2	-
Lagged 3 years	of three years previous	ppt_anu_dis3 ppt_spr_dis3 ppt_sum_dis3 ppt_fal_dis3 ppt_win_dis3	tem_anu_dis3 tem_spr_dis3 tem_sum_dis3 tem_fal_dis3 tem_win_dis3	-
2 years combined	Of same year summed with previous year (precipitation) or same year averaged with previous year (temperature)	ppt_anu_2 ppt_spr_2 ppt_sum_2 ppt_fal_2 ppt_win_2	tem_anu_2 tem_spr_2 tem_sum_2 tem_fal_2 tem_win_2	-
3 years combined	Of same year averaged with previous 2 years (temperature)	-	tem_anu_3 tem_spr_3 tem_sum_3 tem_fal_3 tem_win_3	-

*Anu = annual, spr = spring, sum = summer, fal = fall, win = winter

Simple regressions were conducted for all variables with a Euclidian distance value <1 to help identify the nature of the relationship between the climate variable and tree establishment numbers; however, only significant regressions with the strongest correlations were reported.

Livestock grazing

Data summarizing livestock grazing were obtained from CHIP-AB records, government publications, and Range Management Plans (Moore 1979; Western Ecological Services Ltd. 1980). Because three stock associations (Medicine Lodge, Battle Creek, and Fox Stock) graze cattle within the study area, data describing grazing in Animal Unit Months (AUM), or the amount of forage required to sustain one "animal unit" for one month, were obtained by range allotments for the years 1957 to 2007. These grazing records covered most of the Alberta portion of CHIP plateau, but also included 108 km^2 of rangeland located outside of the study area (Figure 2).

During the time period 1957 to 2007, grazing records were available in two different formats. From 1957 to 1976, grazing was reported by AUM allocated, which represented the maximum potential stocking of an allotment based on the carrying capacity of the rangeland, with annual changes depending on rangeland health for that year (Western Ecological Services Ltd. 1980). For the years 1977 to 2007, records indicating AUM-used were available, which represented the amount of actual grazing in an allotment between June 1 and October 15 of that year. AUM-used were usually less than the AUM-allocated for all stock associations between 1977 and 2007. Because AUM-used was a better indicator of actual grazing intensity than those allocated,

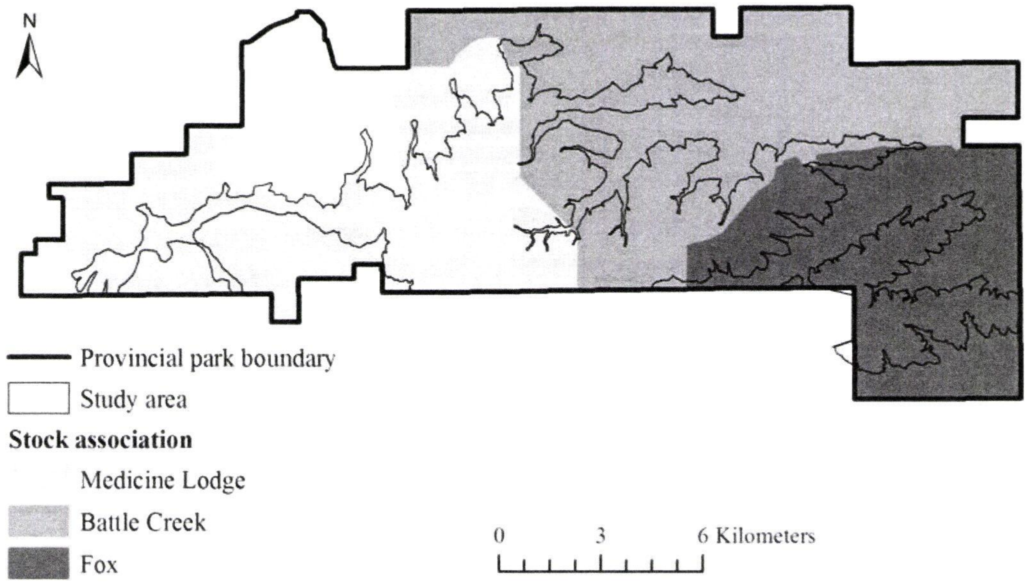

Figure 2. Location of range allotments by stock association in Cypress Hills Interprovincial Park – Alberta with respect to the study area. Continuous line within park represents study area boundary.

AUM-used were approximated from those allocated for the time period 1957 to 1976 for each stock association. This approximation was accomplished by determining percent of AUM-used relative to those that were allocated each year for each association from 1977 to 2007. AUM-allocated for each association from 1957 to 1976 were then prorated by these respective percentages to estimate AUM-used.

Grazing records were compared against tree establishment numbers. To smooth the data and identify trends, five-year running averages of both data sets were used. Trees under five years of age were excluded from the analysis, because numbers of seedlings may not necessarily reflect the number of surviving trees over time, as was the approach taken by Hessl and Baker (1997). Consequently, livestock grazing records were reported from 1959 to 2005, and tree establishment data were reported from 1940 to 2001. The additional time period reported for tree establishment numbers allowed the identification of trends prior to changes in grazing records.

Elk grazing

To determine the potential relationship between elk grazing within forest and tree encroachment areas, elk population numbers from yearly aerial surveys conducted by the Government of Alberta between 1978 and 2006 were converted to cattle-equivalent AUM to maximize their comparability. Elk consume 67% of the forage utilized by cattle (Heady 1975), therefore, the following equation was used to estimate yearly cattle-equivalent AUM-used by elk:

Equation 6: $$\text{Yearly Elk AUM-used [cattle-equivalent]} = \text{Elk population size} \times 0.67 \times 12 \text{ months}$$

A linear regression model was used to estimate AUM-used by elk between 1937 and 1978, with AUM equal to zero in 1937. Five-year running averages of AUM-used by elk and tree establishment numbers were analyzed, with trees less than five years of age excluded. Elk population numbers were reported from 1959 to 2005, and tree establishment data were reported from 1940 to 2001.

CHAPTER 3: RESULTS

Field work occurred between May 1 and August 13, 2007, and 109 transects were sampled. One hundred transects were located on the perimeter of the Cypress Hills plateau and extended from the 1950 tree line or plateau margin into grassland, six were located across forest islands, and three occurred in grassland areas. Transects ranged from 50 to 890 m in length, and had a combined length of 24.1 km. The average transect length was 221 m, but median and mode values were 200 m. The three transects located in grassland were 200 m in length. Over half (55%) of the transects on the perimeter of the plateau were 200 m in length, whereas 25% were shorter and 20% were longer. Transects through forest islands were longer, with an average length of 387 m. Five transects did not encounter any trees. A total of 1,361 trees were cored or clipped. *Picea albertiana, Pinus contorta, Populus balsamea,* and *Populus tremuloides* trees were sampled. As only one *Populus balsamea* was encountered, this species was omitted from the analysis.

Tree sample characteristics

Pinus contorta comprised almost half of all sampled trees, and *Populus tremuloides* comprised 68% of the remaining samples (Table 3). Most samples occurred in the ≥50 mm and <25 mm basal diameter size-classes. The <25 mm size-class was dominated by *Populus tremuloides,* whereas *Pinus contorta* was the dominant species of the >25 mm size-classes.

Table 3. Number of sampled trees by species and size-class.

Size-class	*Picea albertiana* Number of samples	Percent	*Pinus contorta* Number of samples	Percent	*Populus tremuloides* Number of samples	Percent	Total
<20 mm	90	40.0	88	13.5	357	73.8	535
20-49 mm	44	19.6	80	12.3	16	3.3	140
≥50 mm	91	40.4	483	74.2	111	22.9	685
Total	225		651		484		1360

Trees ranged from 1 to 143 years old (Table 4). The oldest tree was a *Pinus contorta*; however, *Populus tremuloides* samples as a group were significantly (P <0.001) older than either *Pinus contorta* or *Picea albertiana* in the ≥50 mm size-class (Table 4). *Picea albertiana* samples were older than the other two species in the 25-49 mm size-class, whereas *Populus tremuloides* samples were younger in the <25 mm size-class. The greatest range in mean ages among size-classes occurred in *Populus tremuloides* samples (60 years), but was smallest among *Picea albertiana* samples (24 years). The least amount of variability in age occurred in the <25 mm size-class, and increased with increasing basal diameter across all species (Table 4).

Figure 3 illustrates the frequency of tree establishment by species and year regardless of basal diameter size-class. The number of trees established on the plateau increased at an exponential rate after 1893 (Figure 4). All trees were established after this year except the oldest tree. *Pinus contorta* and *Populus tremuloides* were both present in the study area prior to 1907, whereas *Picea albertiana* first occurred within the data between 1908 and 1917. Establishment numbers of both coniferous species increased slightly from 1908 to 1977. A greater increase occurred for both species in the subsequent 20 years, with 51% and 63% of individuals established.

Table 4. Number and age characteristics of the sampled trees based on basal diameter size-class. Species ages within size-classes were compared using Kruskal-Wallis tests with differences identified using Scheffé rank tests.

		Basal diameter size-class								
			<20 mm			20-49 mm			≥50 mm	
Species		PIAL*	PICO	POTR	PIAL	PICO	POTR	PIAL	PICO	POTR
Number of Samples		90	88	357	44	80	16	91	483	111
Tree Age	Minimum	2	2	1	2	8	7	9	8	22
	Maximum	30	23	14	57	30	25	90	143	119
	Mean	12	12	4	23	16	14	36	45	64
	Standard deviation	5	4	2	10	6	5	18	28	18
	Kruskal-Wallis test (*P*)		<0.001			<0.001			<0.001	
	Scheffé rank†	b	b	a	b	a	a	a	a	B

* PIAL = *Picea albertiana*, PICO = *Pinus contorta*, and POTR = *Populus tremuloides*.
† The letters show Scheffé rank test results. Different letters indicate which species were significantly different in age.

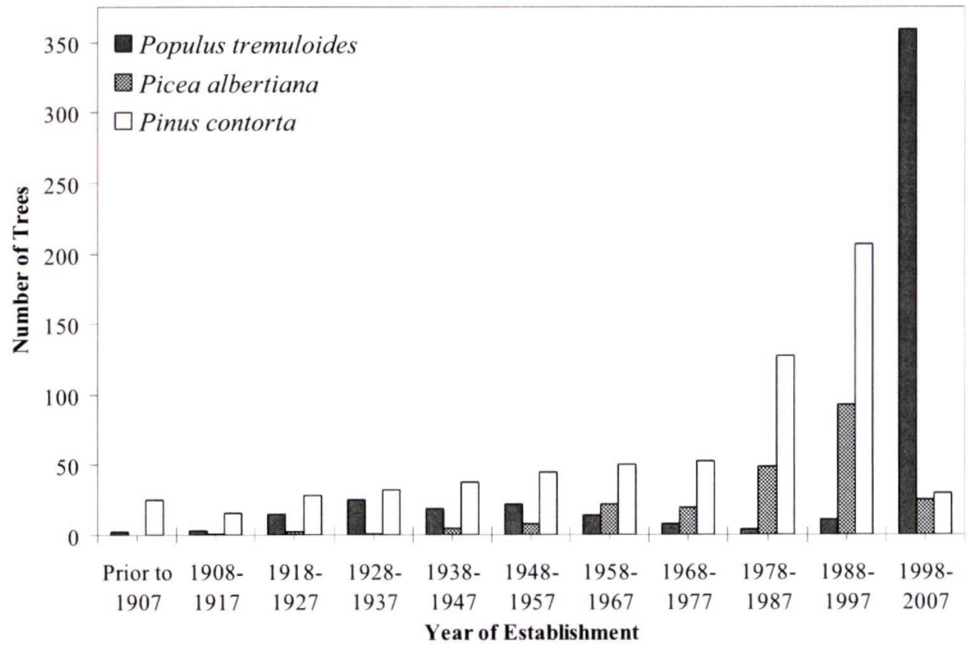

Figure 3. Time period of tree establishment by species based on 1,358 samples.

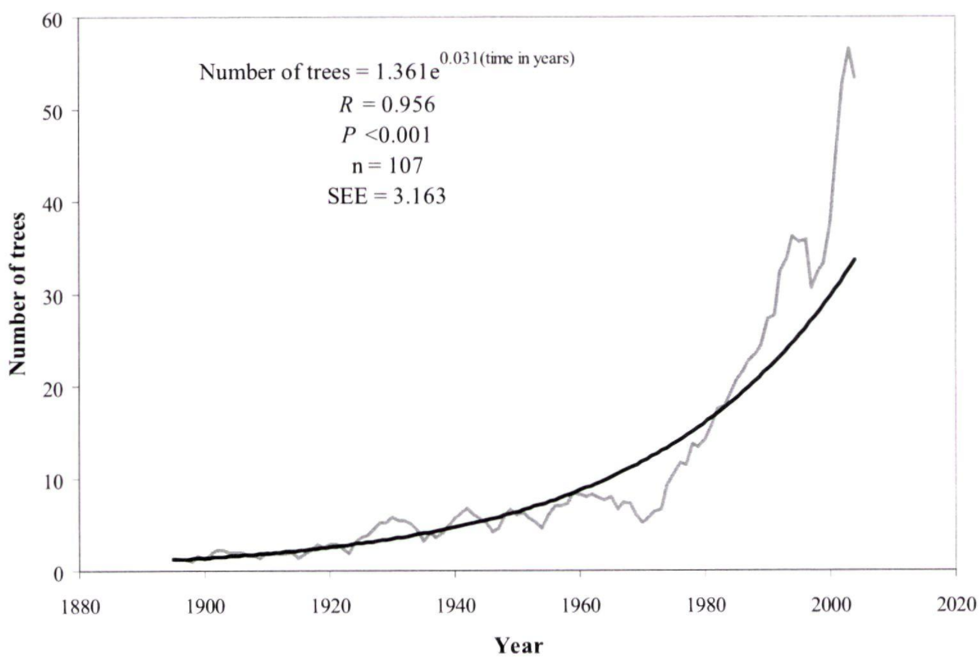

Figure 4. Establishment numbers of all sampled trees (grey line) from 1893 to 2005 based on 5-year averages, with exponential regression model (black line).

all sampled *Pinus contorta* and *Picea albertiana*, respectively. In contrast, 75% of *Populus tremuloides* established during the last decade. Prior to 1998, *Populus tremuloides* did not exhibit a pattern of continual increase, but rather a weak cyclic pattern, with a peak in the 1928 to 1937 period (Figure 3).

Figures 5, 6, and 7 show tree establishment frequencies by decade according to basal diameter age-class. *Pinus contorta* frequencies increased across all three size-classes, except in the most recent decade, with the greatest numbers occurring in the ≥50 mm size-class between 1978 and 1987. The greatest establishment of *Picea albertiana* occurred among samples with a basal diameter of <25 mm between 1988 and 1997. *Populus tremuloides* establishment did not increase notably after 1937 in the ≥50 mm size-class, and appeared to decrease after 1957. The greatest number of established *Populus tremuloides* occurred after 1998 in the <25 mm size-class (Figure 7).

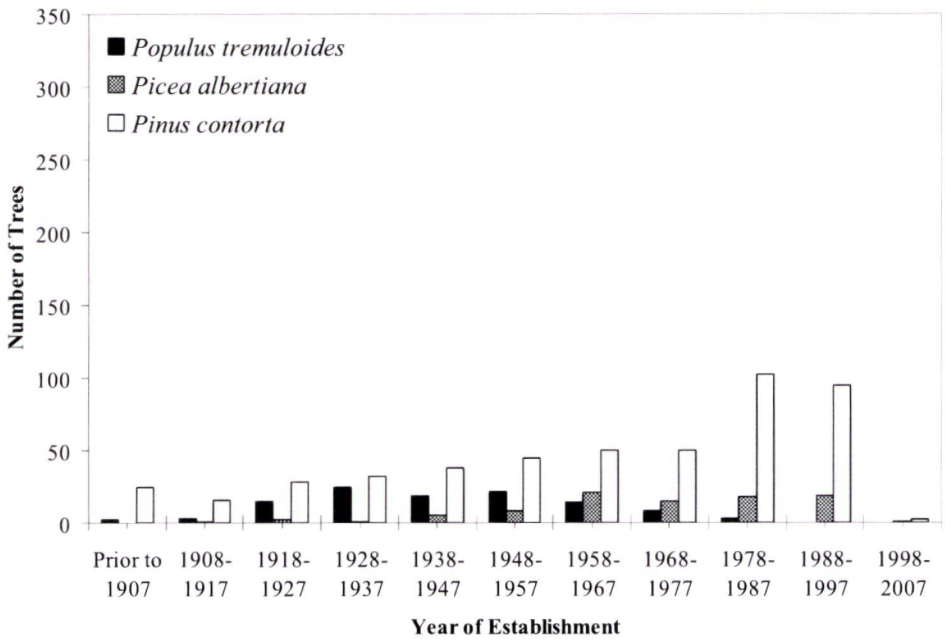

Figure 5. Time period of tree species establishment with a basal diameter ≥50 mm based on 685 samples.

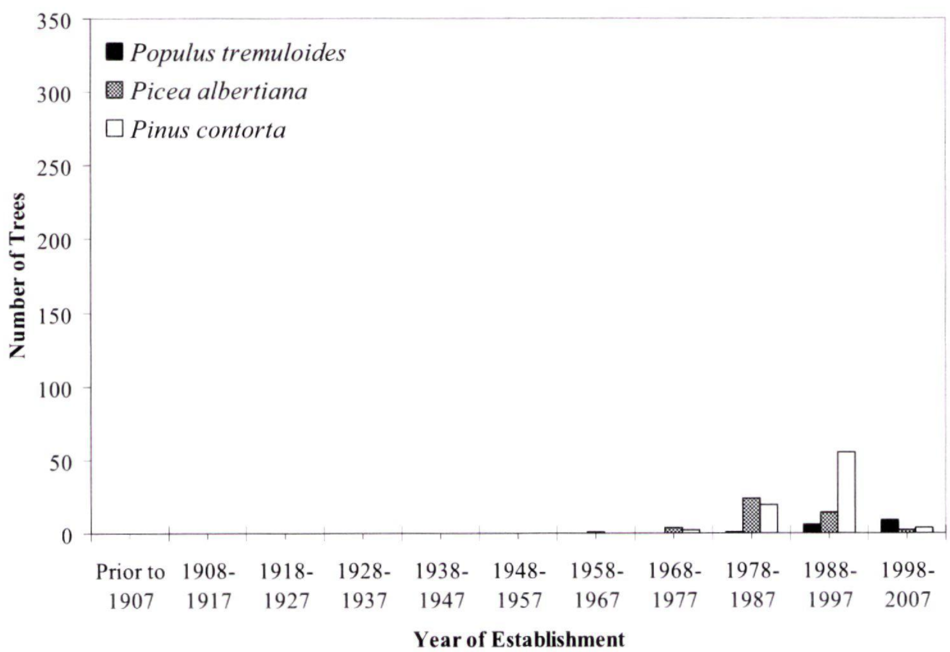

Figure 6. Time period of tree species establishment with a 25-49 mm basal diameter based on 140 samples.

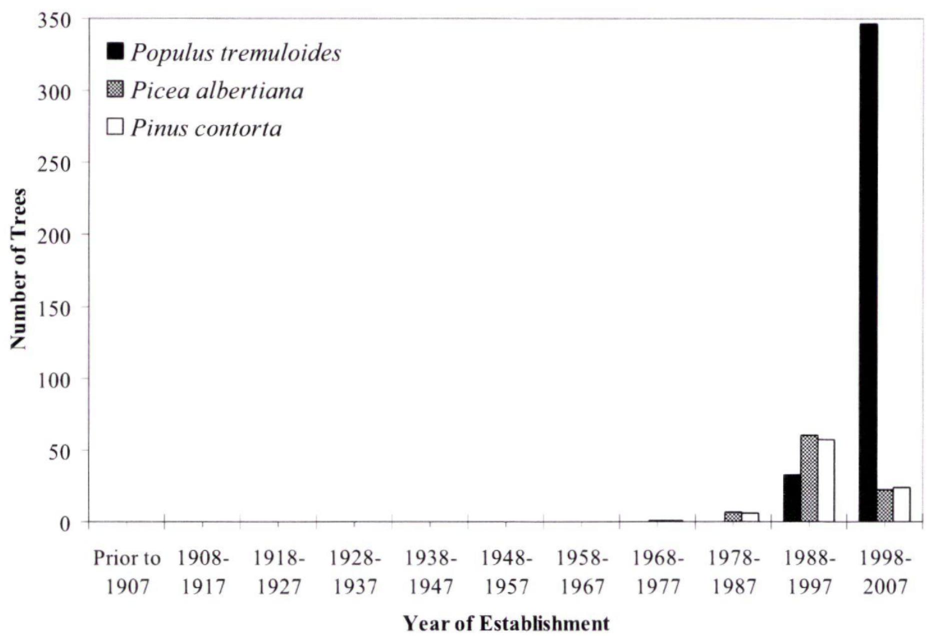

Figure 7. Time period of establishment of tree seedlings with a basal diameter <25 mm based on 535 samples.

Pre-1950 forest and isolated trees

A total of 1,502 ha of forest and tree patches were present on the CHIP-AB plateau prior to 1950 based on air photo analysis (Figure 8). Most forest was located along the perimeter of the plateau, but extended to over a kilometer from the edge onto the surface in some locations. Isolated trees and forest islands also occurred, especially near dry valleys.

Samples collected from pre-1950 forests (n = 476) revealed that it was composed of *Pinus contorta*, *Populus tremuloides,* and *Picea albertiana* (Figure 9). Among these samples, 203 trees established prior to 1950, of which *Pinus contorta* comprised 60% and *Populus tremuloides* 35%. Establishment of both these species increased after 1908, with

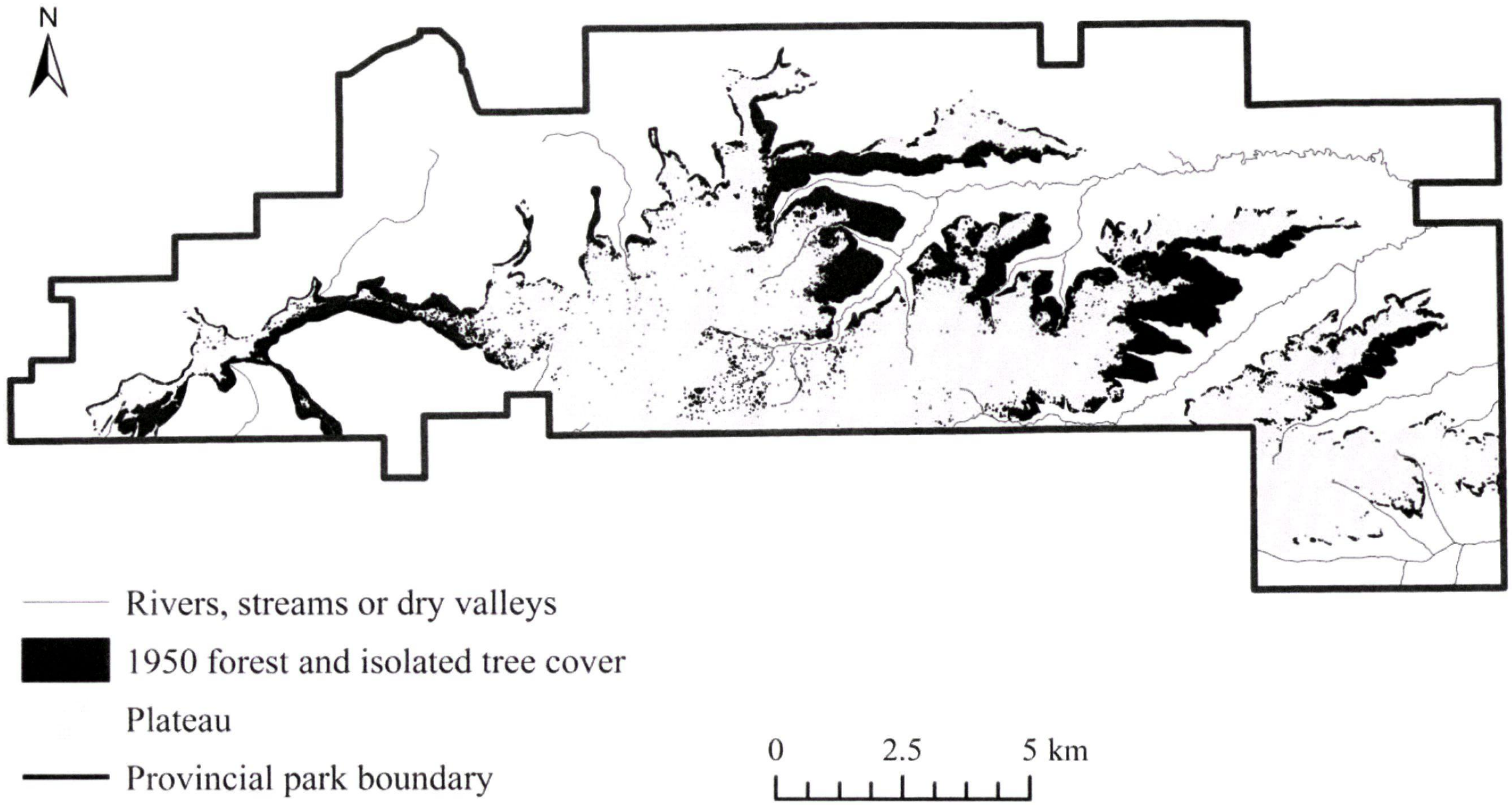

Figure 8. Forest and isolated tree areas on the Alberta portion of the Cypress Hills plateau in 1950.

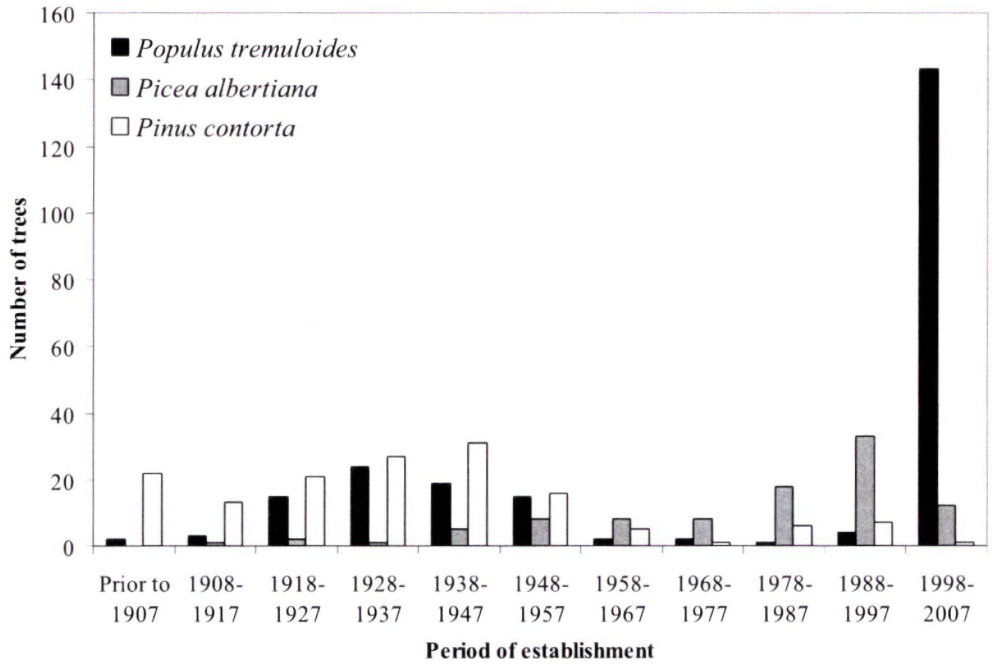

Figure 9. Period of tree establishment by species in pre-1950 forests based on 476 samples.

Pinus contorta peaking between 1938 and 1947, and except for the last decade, *Populus tremuloides* peaking between 1928 and 1937. A large increase in *Populus tremuloides* frequencies occurred during the last decade. In contrast, the majority (82%) of *Picea albertiana* in the pre-1950 forests were established after 1950. *Picea albertiana* followed an overall pattern of increase from 1908 to 1997.

Pre-1950 forests were dominated by *Pinus contorta*; however, pre-1950 forest stands dominated by *Populus tremuloides* occurred on 17 transects, usually between pre-1950 *Pinus contorta* forest and grassland vegetation. The *Populus tremuloides* in these stands were often rotten in the center, wind blown, stunted, or had sparse foliage, i.e., generally of poor health. Only two *Pinus contorta* trees established after 1950 within these stands, whereas *Picea albertiana* establishment increased noticeably from 1958 to 1997 (Figure 10).

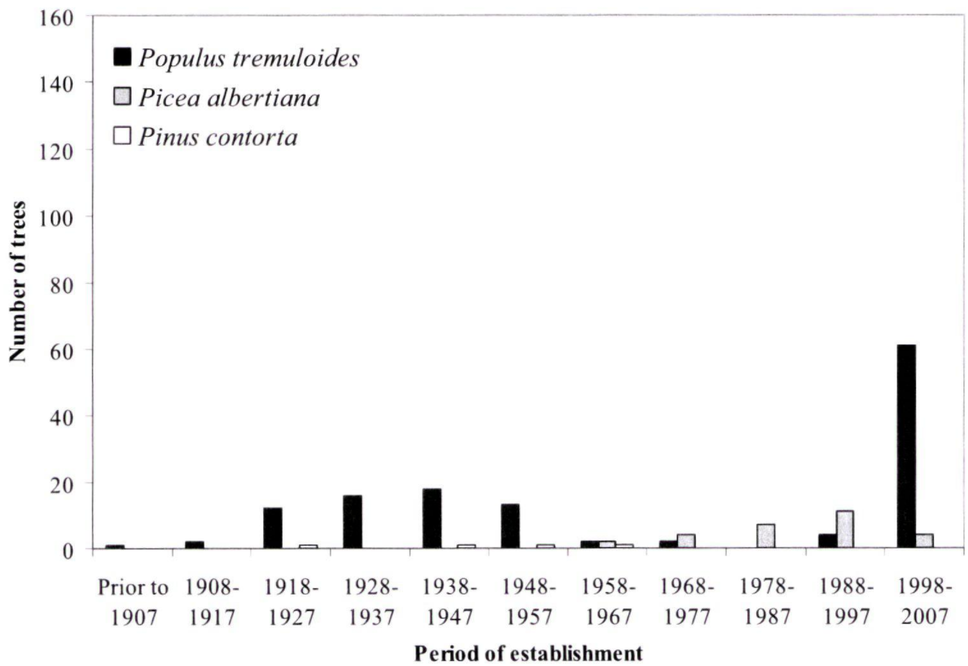

Figure 10. Period of tree establishment by species in pre-1950 *Populus tremuloides*-dominated stands based on 163 samples.

Encroachment between 1951 and 2002

Temporal and spatial analysis of remotely-sensed images revealed that forest and isolated tree areas on the CHIP-AB plateau increased from 1,502 ha in 1950 to 2,270 ha in 2002; a 51% increase over 52 years (Figure 11; Table 5). This amount of increase represents 10% of the plateau surface. New forest amounted to 763 ha, with an additional 7 ha representing isolated tree patches. The rate of increase, or encroachment, was relatively constant with a slight increase between 1992 and 2002 (Figure 12).

Table 5. Forest and isolated tree areas on the Cypress Hills Interprovincial Park – Alberta plateau from 1950 to 2002 based on remotely-sensed image analysis. These area estimates to not include the area of tree ingress into grassland.

	1950	1951-1971	1972-1981	1982-1992	1993-2002	Increase
Forest area (ha)	1,492	--	--	--	--	--
Isolated tree area (ha)	8	10	12	12	15	7
New forest area (ha)	--	243	117	171	232	763
Total area (ha) (cumulative)	1,502	1,745	1,864	2,035	2,270	770
Percent increase	--	16	7	9	12	51

Average rate of encroachment according to the linear regression model was ~14.5 ha of new forest and isolated trees per year based on Table 5 values (Figure 12). Appendix C provides forest cover maps resulting from the spatial analysis of each air photo set.

Between 1951 and 1971, 243 ha of new forest developed on the CHIP-AB plateau (Table 5). Most developed along the 1950 tree line and around existing forest islands. Some new forest islands were present in 1971, which were largely located near dry valleys. Isolated tree occurrences increased by 2 ha during this 21 year period, also near dry valleys, and in some cases, centrally on the plateau. New forest in the amount of 117 ha developed between 1972 and 1981, and isolated tree coverage increased by 2 ha (Table 5). Between 1982 and 1992, 171 ha of new forest developed, but isolated tree coverage did not change (Table 5). During this 21-year period, less forest developed perpendicular to the plateau edge. Rather development was concentrated in previously treeless patches surrounded by forest (i.e., infilling), both along the perimeter and

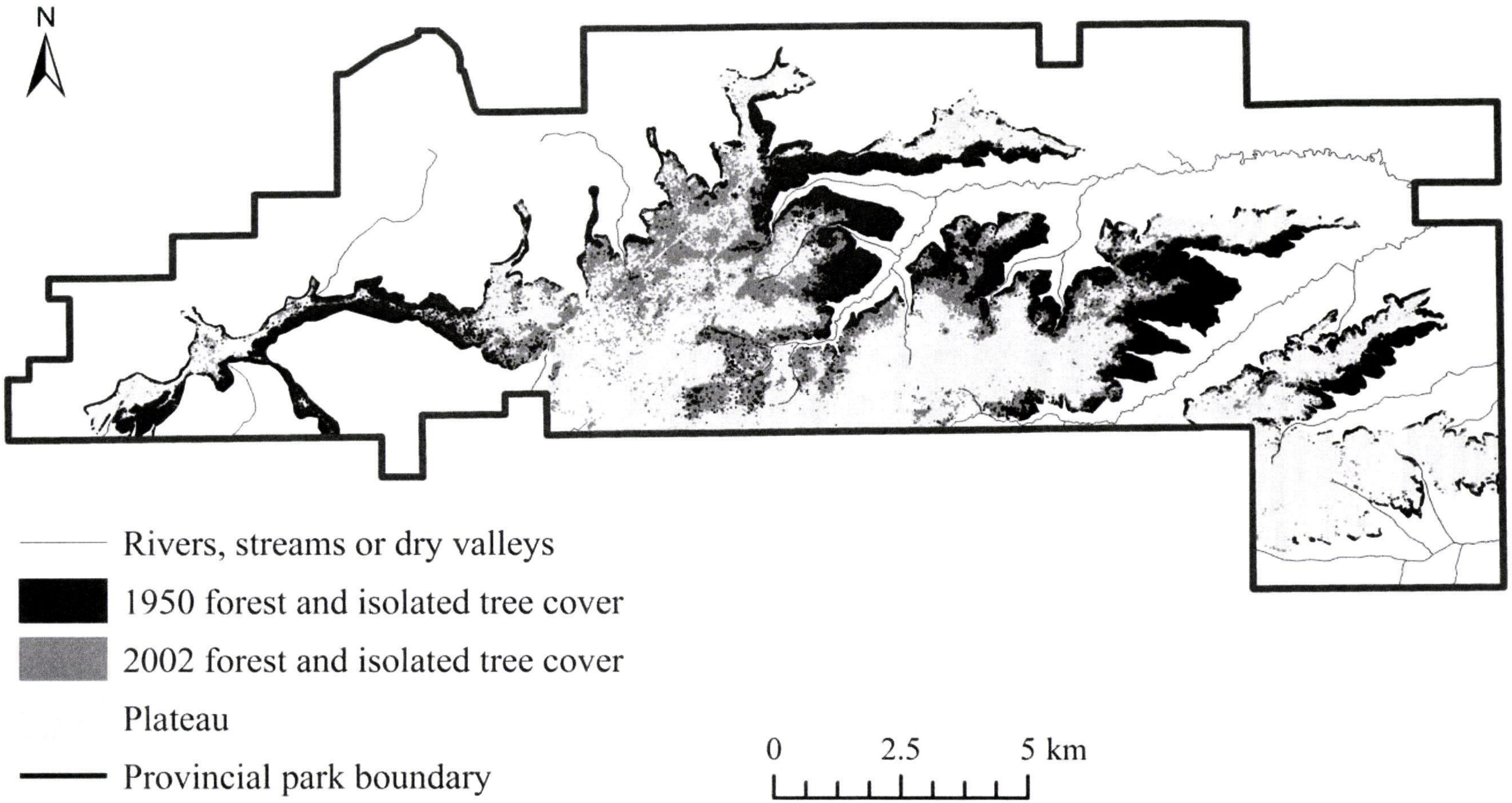

Figure 11. Forest and isolated tree area increase on the Alberta portion of the Cypress Hills plateau between 1950 and 2002.

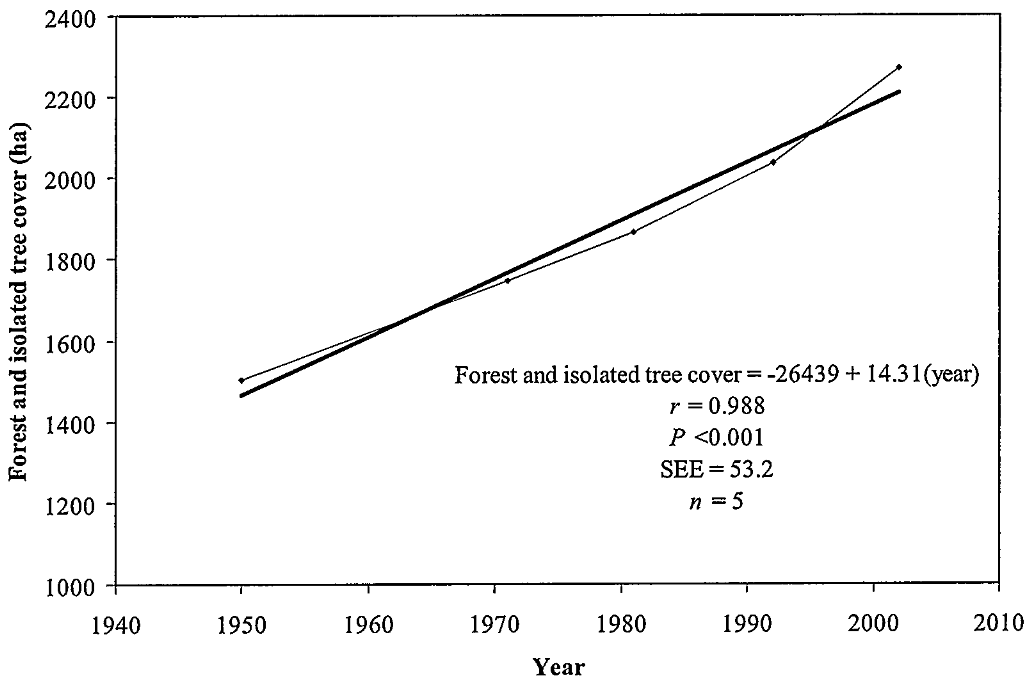

Figure 12. Rate of forest and tree encroachment (dotted line) on the Cypress Hills Interprovincial Park – Alberta plateau from 1950 to 2002, with linear regression model (solid line)

between forest islands. Expansion of new forest and trees into grassland around forest islands was apparent, again mostly near dry valleys.

Between 1992 and 2002, 232 ha of new forest encroachment occurred (Table 5). Isolated trees increased by 3 ha. Forest expansion perpendicular to the edge of the plateau resumed, and expansion around and between forest islands continued. New tree development, especially in the south-central third of the study area, was more apparent during this decade than in previous periods.

Overall, between 1951 and 2002, a large portion of new forest and trees were established in the central third of the study area (Figure 11). Isolated trees were present in the north-central third in 1950; however, it was otherwise devoid of forest (Figures 8). Forest islands had developed by 1971, and they expanded and converged during the subsequent 30 years. In the remainder of the study area, new forest was concentrated along the perimeter of the plateau, filling spaces devoid of tree cover, and expanding perpendicular to the 1950 tree line.

Ground truthing suggested that the forest and tree cover in 2002, determined by spatial analysis, represented forest and tree areas on the ground in 2007 with an acceptable degree of accuracy. Average error of spatial analysis results compared with data collected in the field was 3%. Of the trees sampled in the $\geq$50 mm size-class (n = 686), 99% were located on sample points that intersected a forest or isolated tree polygon. Of the sample points occurring in grassland vegetation in the field (n = 1,303), 96% did not intersect a forest or isolated tree polygon.

New forest and tree encroachment in 2007

Total encroachment (post-1950 forest and areas of tree ingress) was estimated to represent 1,531 ha, or 20% of the plateau in 2007, based on calculations using Equation 2. New forest was estimated to cover 780 ha in 2007 by extrapolation from the linear rate of encroachment in Figure 12. This was an expansion of 52% since 1950, and the area of increase covered 11% of the plateau. The balance of the encroachment area therefore indicated that 768 ha of tree ingress area (isolated trees and surrounding grassland) existed in 2007, occupying 9% of the plateau.

Figure 13 shows that tree establishment numbers began to increase substantially in post-1950 forest and ingress areas after 1940, and aside from two periods of decrease from 1965 to 1970 and from 1995 to 2000, increased steadily until 2005. The rate of tree establishment also accelerated several-fold after the mid-1940s (i.e., an exponential rate, see Figure 4). Both *Pinus contorta* and *Picea albertiana* increased from 1948 to 1997 (Figure 14). *Populus tremuloides* establishment numbers increased substantially after 1997. Seven different types of stands, based on the trees with a ≥50 mm basal diameter, composed the encroachment area (Table 6). Stands dominated by *Pinus contorta* accounted for most of the encroachment (90%), and pure *Pinus contorta* was the most common stand type (Table 6). *Picea albertiana* was present in 32% of new stands, mostly in combination with *Pinus contorta. Populus tremuloides* occurred in <8% of stands.

Of the 53 trees sampled within the area of ingress on the plateau, *Pinus contorta* comprised 62%. Five trees were *Picea albertiana* (Table 7). All trees were established after 1950. Over half (55%) of these trees were >50 mm in basal diameter, whereas 40% were <25 mm in basal diameter, occurring at densities of 54 and 55 stems per hectare, respectively. The largest range in ages occurred in *Pinus contorta* trees. All *Populus tremuloides* were <10 years old and *Picea albertiana* ages ranged from 30–50 years.

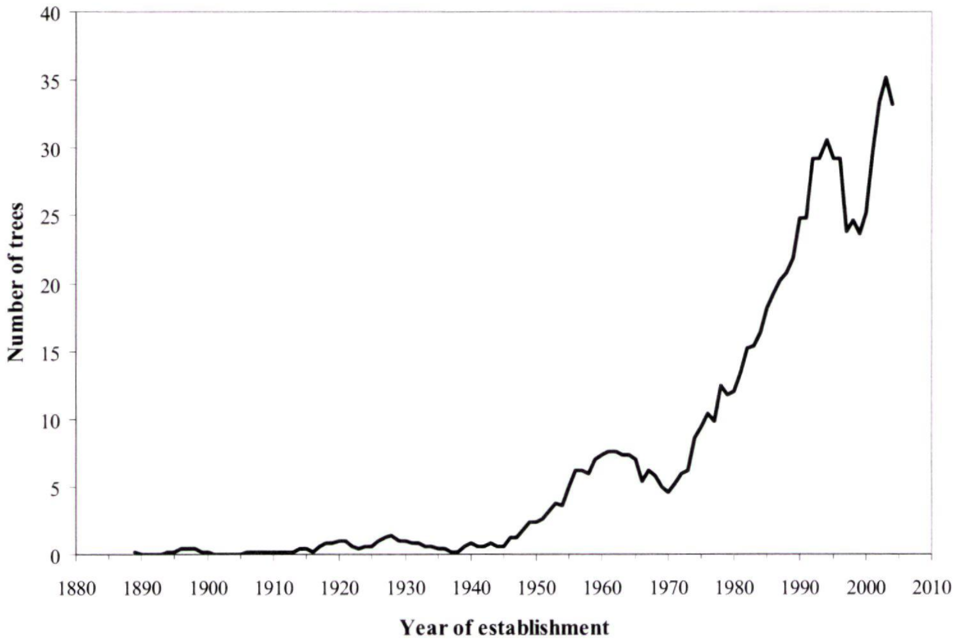

Figure 13. Tree establishment numbers within the post-1950 area of encroachment based on 5-year averages.

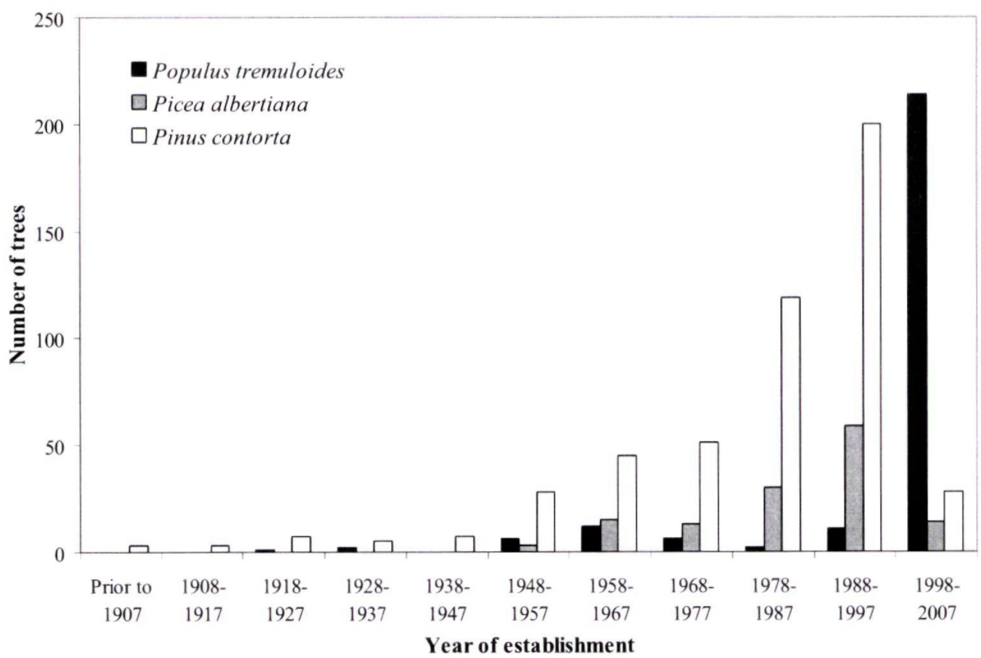

Figure 14. Time period of tree establishment by species within the area of encroachment based on 882 samples.

Table 6. Stand type and associated area, based on species of trees >50 mm in basal diameter, within the encroachment area on the plateau.

Stand type	Number of transects	Average density of trees (trees/ha)	Total (ha)	Percent of total
Pinus contorta	23	377	586	62
Pinus contorta - Picea albertiana	7	185	252	27
Populus tremuloides – Picea albertiana	5	658	38	4
Picea albertiana	7	227	25	3
Pinus contorta – Picea albertiana – Populus tremuloides	2	225	25	3
Populus tremuloides	3	114	9	<1
Pinus contorta – Populus tremuloides	2	1510	4	<1
All types	49	364	940	100

Table 7. Number and age characteristics of sampled trees within the area of ingress on the Cypress Hills Interprovincial Park – Alberta plateau.

Species	Number of trees	Age (years)			
		Maximum	Minimum	Mean	Standard deviation
Pinus contorta	33	56	6	24	15
Picea albertiana	5	49	31	34	9
Populus tremuloides	15	9	1	4	3

Health and reproductive status

Of the sampled *Picea albertiana* ($n = 225$), *Pinus contorta* ($n = 651$), and *Populus tremuloides* ($n = 484$), 98%, 91%, and 89%, respectively, were considered to be in fair or good health. Most *Pinus contorta* of poor health (7% of total) occurred in the area of encroachment. Of the *Populus tremuloides* considered to be of poor health (52 trees), most were <10 years old.

Over half (60%) of *Pinus contorta* were reproducing based on the presence of cones, but only 30% of *Picea albertiana* had cones. The youngest reproducing individual of both coniferous species was nine years old, and 19% of reproducing *Pinus contorta* were under the age of 30, whereas 10% of reproducing *Picea albertiana* were under this age.

Site characteristics

Ah-horizon thickness

Mean Ah-horizon thickness varied significantly with vegetation type (Table 8). The thickest horizons occurred in areas with grassland vegetation, whereas pre-1950 forests had Ah-horizons that were on average 3.7 cm or about 30% thinner. Areas that became forested or treed after 1950 had significantly thinner Ah-horizons than areas that remained grassland.

Herbaceous and shrub cover

Herbaceous plant cover in pre-1950 forest, tree encroachment, and grassland areas averaged 79%, 88%, and 98%, respectively, and was found to differ significantly among all vegetation types (Table 8). Shrub cover also differed among all vegetation types. Areas of encroachment contained the greatest shrub cover, whereas pre-1950 forest areas contained the least (Table 8). Only two shrub taxa occurred in all three vegetation types: *Dasiphora floribunda* and *Rosa* spp. (wild roses) (Table 9). A significant difference in *Dasiphora* cover occurred among cover types, with most occurring in grassland areas and the least in pre-1950 forest areas. *Rosa* spp. cover was significantly greater in pre-1950 forest than in either areas of encroachment or grassland. *Arctostaphylos uva-ursi* (bearberry), and *Juniperus* spp (junipers) were common to pre-1950 forest and encroachment areas. *Betula pumila* (dwarf birch) occurred in both grassland and encroachment areas.

Slope and aspect

Slope and aspect data did not vary sufficiently among forest and tree encroachment areas to determine if these site conditions had an effect on tree establishment numbers. Of the sample sites that occurred in new forest and areas of ingress, 85% had slope between 0–2%.

Table 8. Site characteristics based on vegetation type. Site characteristics within cover types were compared using Kruskal-Wallis tests, with differences identified using Scheffé rank tests.

	Site characteristics								
	Ah-horizon thickness (cm)			Herbaceous plant cover (%)			Shrub cover (%)		
Vegetation type*	PRE	ENCR	GRS	PRE	ENCR	GRS	PRE	ENCR	GRS
Number of sample points	128	286	380	134	286	398	133	283	398
Minimum	1	1.5	1	1	0	40	0	0	0
Maximum	23	23	23	100	100	100	85	80	100
Mean	8.7	11.1	12.4	78.9	88.2	98.1	6.8	21.3	18.5
Standard deviation	4.2	4.1	4.2	28	24	7.2	12.4	19.5	17.1
Kruskal-Wallis test (*P*)		<0.001			<0.001			<0.001	
Scheffé rank†	a	b	c	a	b	c	a	c	b

* PRE = pre-1950 forest, ENCR = encroachment, and GRS = grassland.
† The letters show Scheffé rank test results. Different letters indicate which species were significantly different in age at the α 0.05 level.

Table 9. Average cover of shrubs by vegetation type on the Cypress Hills Interprovincial Park – Alberta plateau. Comparisons based on Kruskal-Wallis tests, with differences identified using Scheffé rank tests.

	Average coverage (%) in sites where shrubs occurred			*P*
Shrub	Pre-1950 forest	Encroachment	Grassland	
Dasiphora floribunda	35a* n = 19	86b n = 162	91c n = 248	0.001
Rosa spp.	34b n = 47	5a n = 43	8a n =31	<0.001
Arctostaphylos uva-ursi	18 n = 4	9 n = 7	0 n = 0	NT†
Juniperus spp.	2 n = 1	0.5 n = 2	0 n = 0	NT
Betula pumila	0 n = 0	0.5 n = 2	0.7 n = 1	NT
Cornus stolonifera	2.6 n = 2	0 n = 0	0 n = 0	NT
Shepherdia canadensis	9 n = 3	0 n = 0	0 n = 0	NT

*Letters show Scheffé rank test results. Different letters indicate which species had significantly different cover at the α 0.05 level.
† NT indicates shrubs that were not tested for differences among cover types due to small sample sizes.

Causal factors

Climatic trends

Mean annual temperatures during the 1929 to 2005 period were always above 0°C, with the lowest (0.5°C) occurring in 1949, and the highest (2.7°C) in 2005. An increase in annual temperatures occurred between November 1929 and October 2005 (Figure 15A). Regression modelling suggested that temperatures increased on the CHIP-AB plateau from 1.44°C to ~1.99°C after 1929, or ~0.55°C. Four warm – cool cycles occurred, with an average length of 19 years. Mean summer temperatures (June to August, inclusive[4]) had a decreasing trend between 1929 and 2005, whereas mean spring (March to May) and winter (December to February) temperatures increased (Figures 15B and 16). Fall temperatures remained constant. Winter temperatures below -13°C didn't occur after 1971, compared with nine occurrences prior to this date.

Annual total precipitation increased between 1929 and 2005 (Figure 17). The lowest recorded precipitation occurred in 1931 at 441 mm, and the highest in 1992 at 685 mm. Based on regression, annual precipitation increased 85 mm on average since 1929. Net annual water balance values (Figure 18) suggest short, low magnitude wet and dry cycles occurred in the study area, with a prominent and extended wet cycle occurring between 1938 and 1958. The most prominent dry periods occurred during 1959–1963 and 1999–2002.

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[4] The year was divided into four seasons of equal length, identical to Jakubos and Romme (1993).

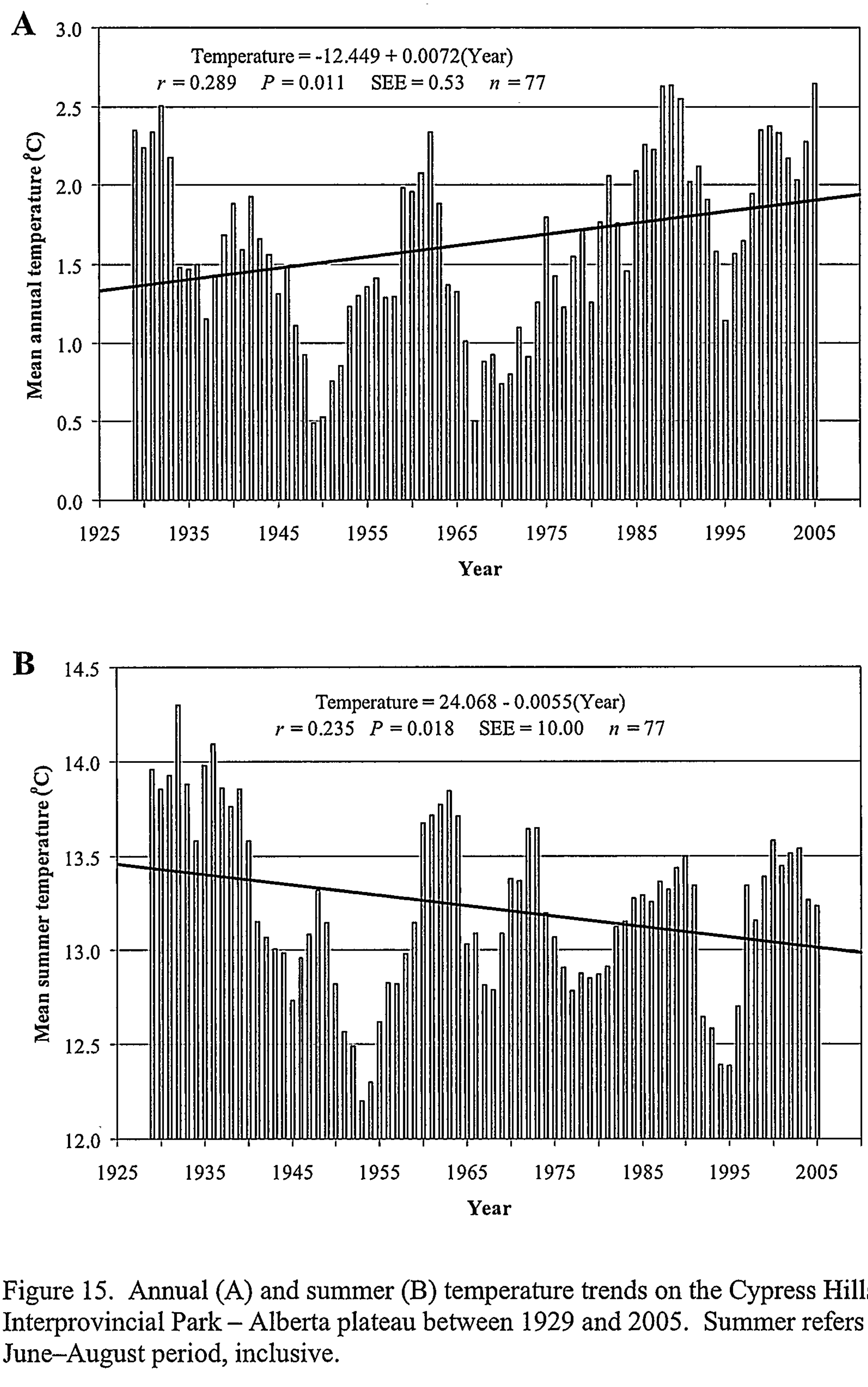

Figure 15. Annual (A) and summer (B) temperature trends on the Cypress Hills Interprovincial Park – Alberta plateau between 1929 and 2005. Summer refers to the June–August period, inclusive.

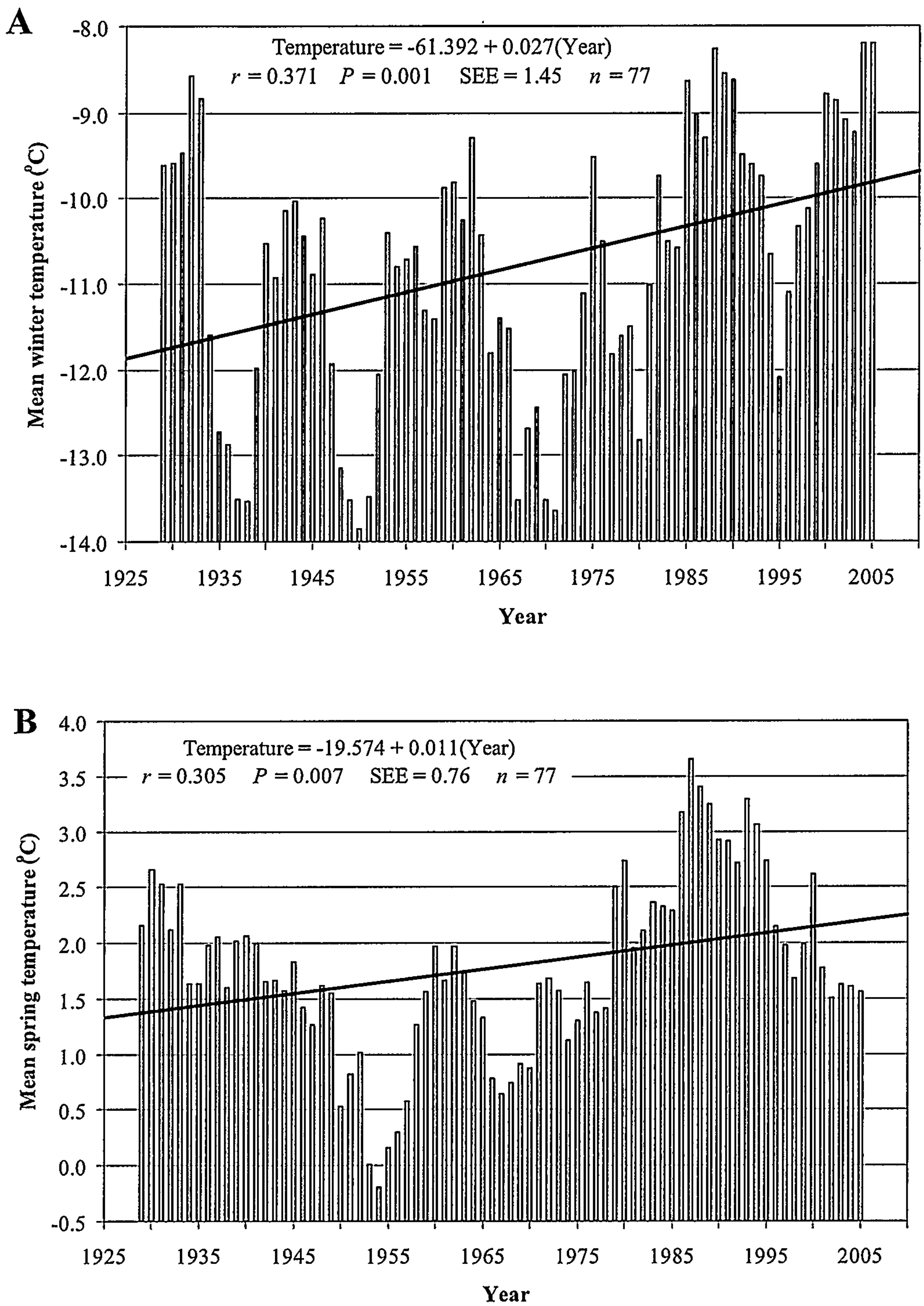

Figure 16. Winter (A) and spring (B) temperature trends on the Cypress Hills Interprovincial Park – Alberta plateau between 1929 and 2005. Winter and spring refer to December–February and March–May period, respectively.

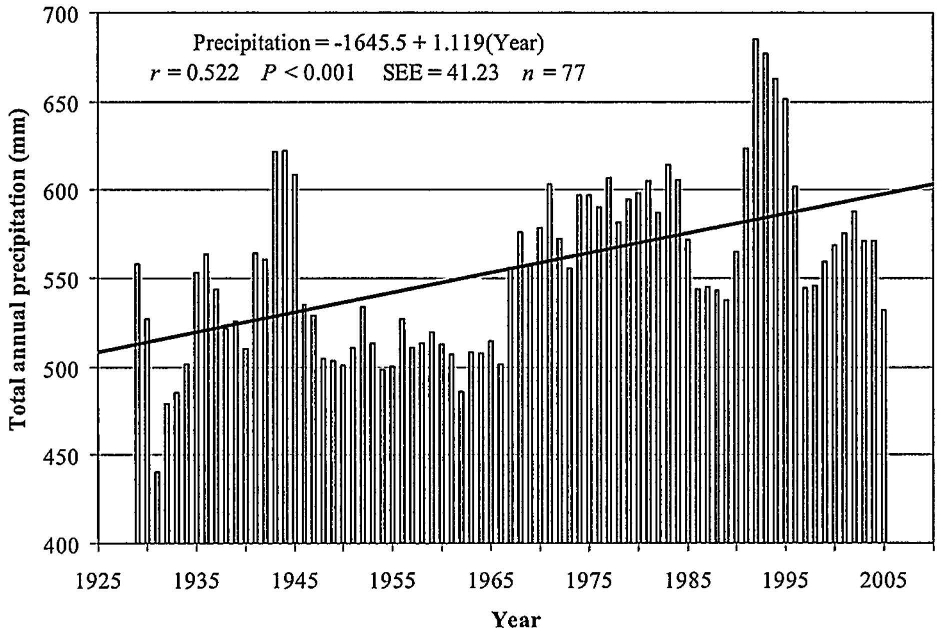

Figure 17. Precipitation trends on the Cypress Hills Interprovincial Park – Alberta plateau between 1929 and 2005, inclusive.

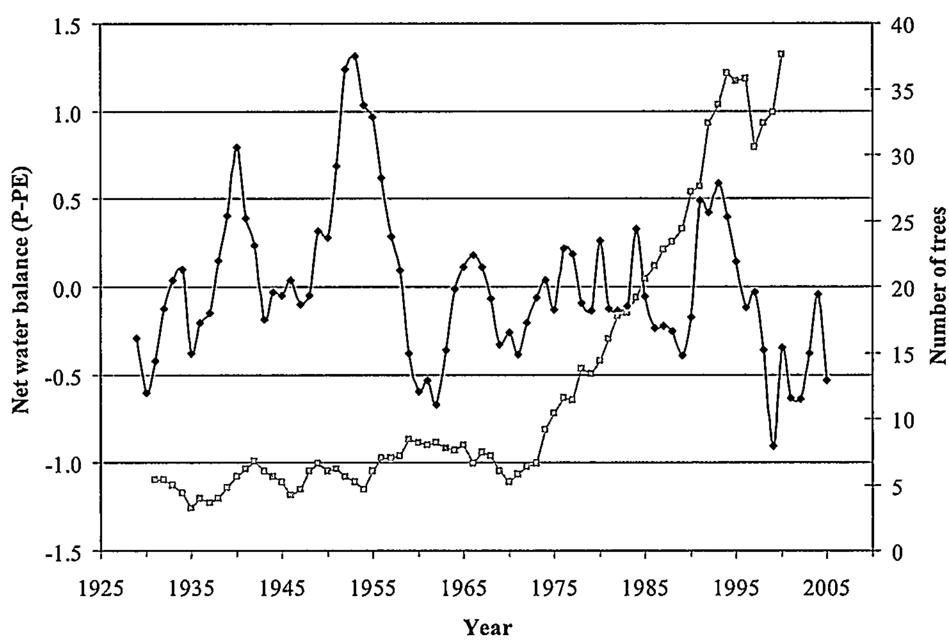

Figure 18. Net annual water balance (P – PE) values (black line) and tree establishment numbers (grey line) on the Cypress Hills Interprovincial Park – AB plateau between 1929 and 2005. Trees 1–5 years old were excluded from the analysis.

Nonmetric multidimensional scaling (MDS) revealed relationships between tree establishment numbers and both precipitation and temperatures variables, but net water balance was not a strong predictor of the establishment of any species (Table 10). The explained variance of the MDS for *Picea albertiana, Pinus contorta* and *Populus tremuloides* was individually 0.970. Variables most closely associated with *Picea albertiana* and *Pinus contorta* were identical, but differed with *Populus tremuloides* (Table 10).

Among precipitation variables, the establishment of all three tree species had a close relationship with fall precipitation (Table 10). Establishment numbers of both coniferous species were linearly related to 5-year averages of fall precipitation (Figures 19 and 20). These two species were also closely related to spring temperature in the MDS (Table 10). Although spring temperature values and spring values lagged by two years had the shortest Euclidean distance (Table 10), 5-year averages of spring temperature had the strongest linear relationship with *Picea albertiana* and *Pinus contorta*, explaining 36% and 43% of variance, respectively (Figures 21 and 22). In contrast, 2-year averages of annual temperature and annual temperature lagged one year were the best predictor variables of *Populus tremuloides* establishment based on Euclidean distances. *Populus tremuloides* establishment was not linearly related to any of the tested variables.

Table 10. Results of nonmetric multidimensional scaling analysis to identify climate variables most strongly associated with *Picea albertiana* (PIAB), *Pinus contorta* (PICO), and *Populus tremuloides* (POTR) establishment. Only variables with a Euclidian distance of <1 between it and one tree species were listed. See Table 2 for a description of variables.

Euclidean distances between tree species and climate variables							
Temperature				Precipitation			
Variable	PIAB	PICO	POTR	Variable	PIAB	PICO	POTR
tem_anu	0.753	1.053	0.646	ppt_fal	0.804	0.674	1.906
tem_spr	0.419	0.459	1.190				
tem_sum	1.331	1.420	0.527				
tem_win	0.694	1.148	1.291				
tem_anu_5	1.006	0.858	0.665	ppt_fal_5	0.543*	0.425	1.731
tem_spr_5	0.510	0.529	1.249				
tem_sum_5	1.553	1.442	0.900				
tem_win_5	1.103	0.778	0.645				
tem_anu_dis1	1.006	1.060	0.433	ppt_fal_dis1	0.979	0.929	1.942
tem_spr_dis1	0.510	0.574	1.103				
tem_sum_dis1	1.553	1.634	0.881				
tem_fal_dis1	1.677	1.697	0.883				
tem_win_dis1	1.103	1.163	0.771				
tem_anu_dis2	0.858	0.917	1.334	ppt_fal_dis2	0.358	0.439	1.380
tem_spr_dis2	0.345	0.400	1.378				
tem_win_dis2	0.969	1.004	0.441				
tem_anu_dis3	0.951	1.001	1.523	ppt_spr_dis3	0.884	0.792	1.665
tem_spr_dis3	0.746	0.842	1.113	ppt_fal_dis3	1.339	1.375	0.503
tem_win_dis3	0.799	0.863	1.228				
tem_anu_2	1.001	1.073	0.328	ppt_fal_2	0.833	0.716	1.961
tem_spr_2	0.464	0.567	1.121				
tem_sum_2	1.429	1.535	0.706				
tem_fal_2	1.663	1.694	0.873				
tem_win_2	1.029	1.099	0.906				
tem_anu_3	0.889	0.975	0.449	Net water balance (P-PE)			
tem_spr_3	0.448	0.552	1.160	P-PE_fal	1.001	0.863	1.984
tem_sum_3	1.428	1.529	0.862				
tem_win_3	0.840	0.912	0.613	P-PE_fal_5	0.844	0.743	1.803

*Double underscores indicate the variables with the lowest Euclidian distance values for each species.

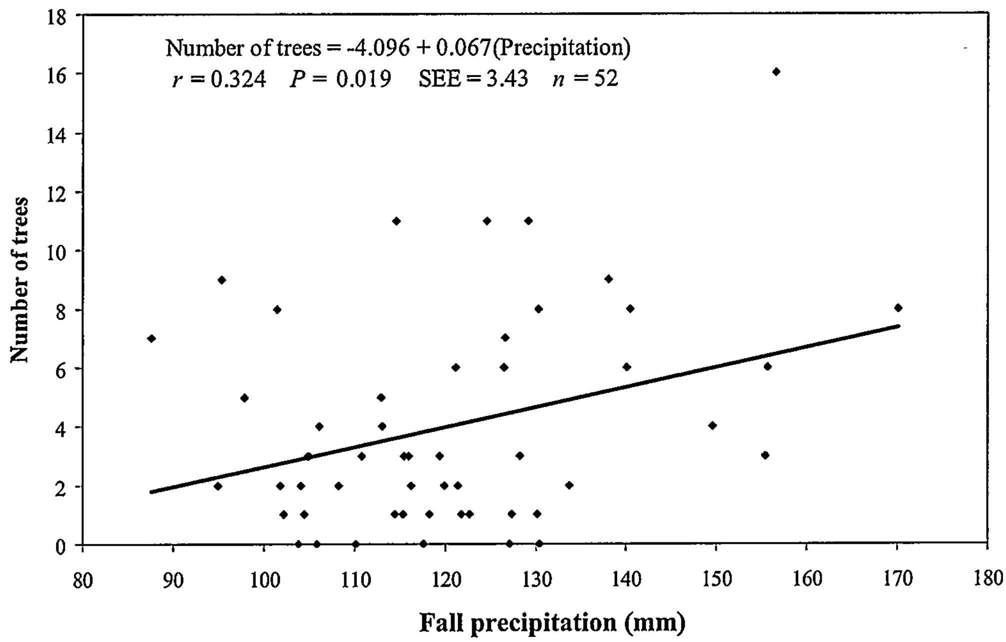

Figure 19. *Picea albertiana* establishment and fall precipitation (5-year average) between 1950 and 2001.

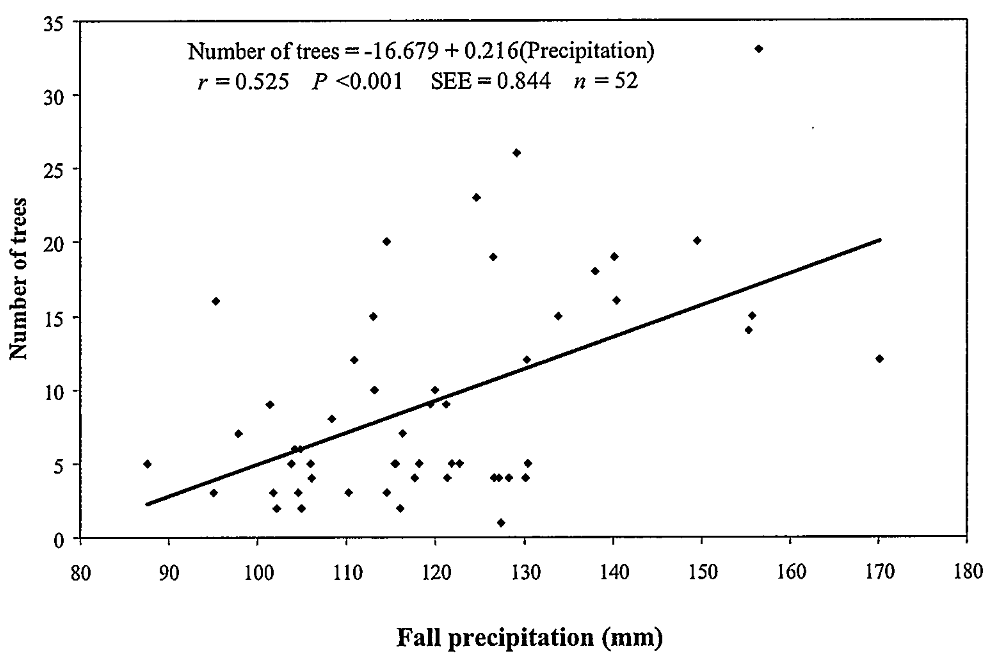

Figure 20. *Pinus contorta* establishment and fall precipitation (5-year average) between 1950 and 2001.

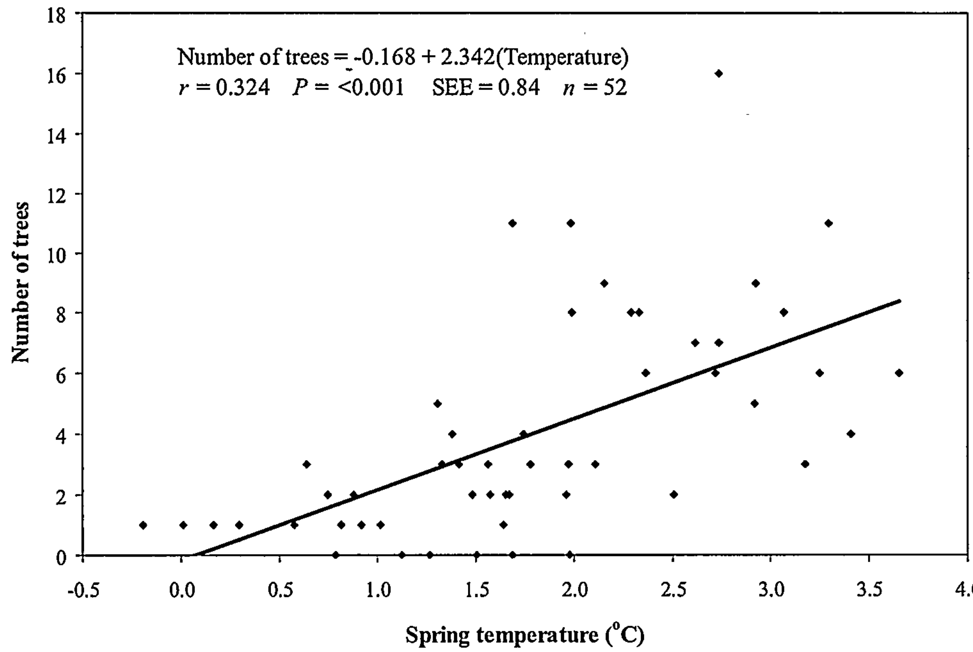

Figure 21. *Picea albertiana* establishment and spring temperature (5-year average) between 1950 and 2001.

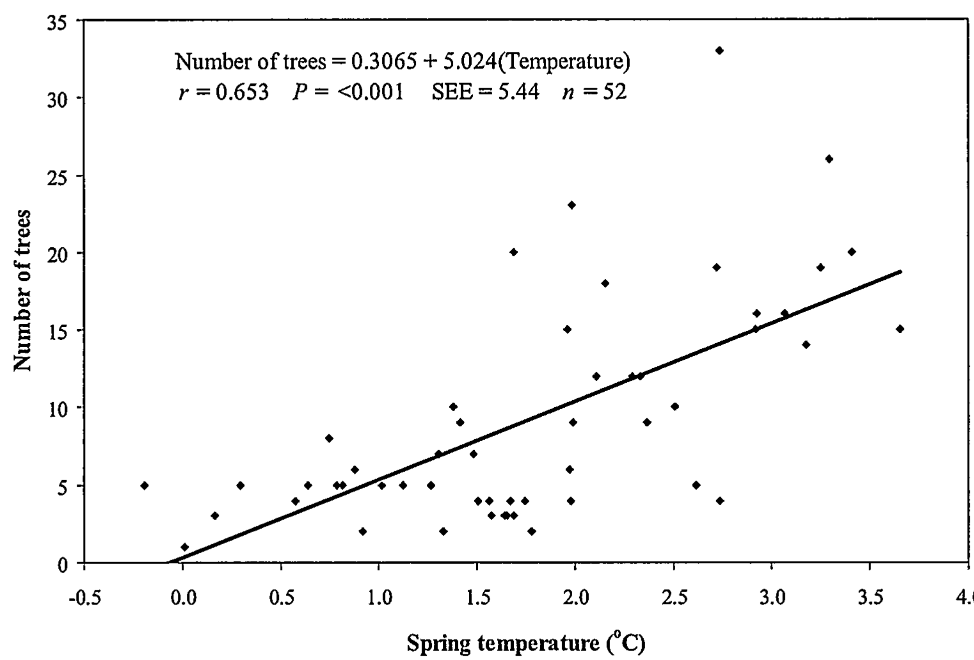

Figure 22. *Pinus contorta* establishment and spring temperature (5-year average) between 1950 and 2001.

Of the climate variables correlated with tree establishment numbers, annual and spring temperature increased during the time period 1950 to 2001 (Figures 23 and 24), whereas fall precipitation remained constant (Figure 25). Spring temperature increased at a greater rate than annual temperature (cf. Figures 24–25).

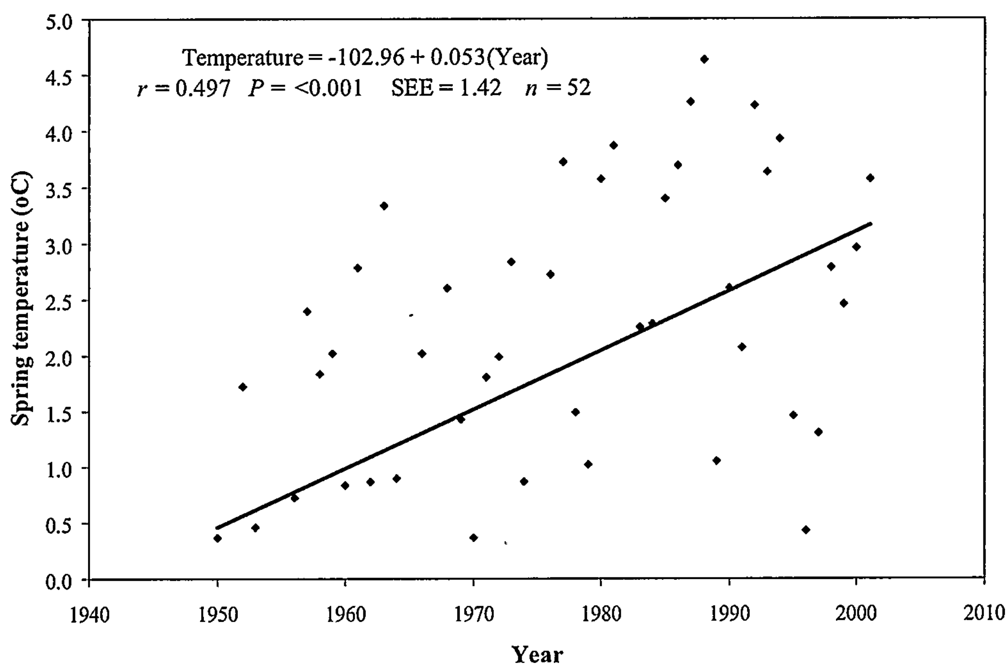

Figure 23. Spring temperature trends on the Cypress Hills Interprovincial Park – Alberta plateau between 1950 and 2001.

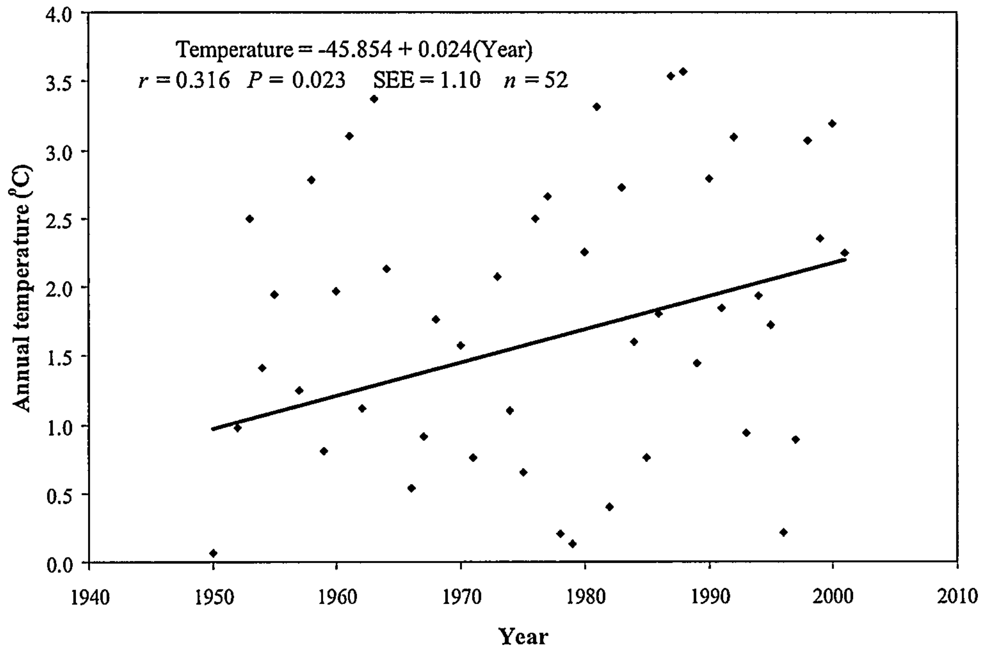

Figure 24. Annual temperature trends (°C) on the Cypress Hills Interprovincial Park – Alberta plateau between 1950 and 2001.

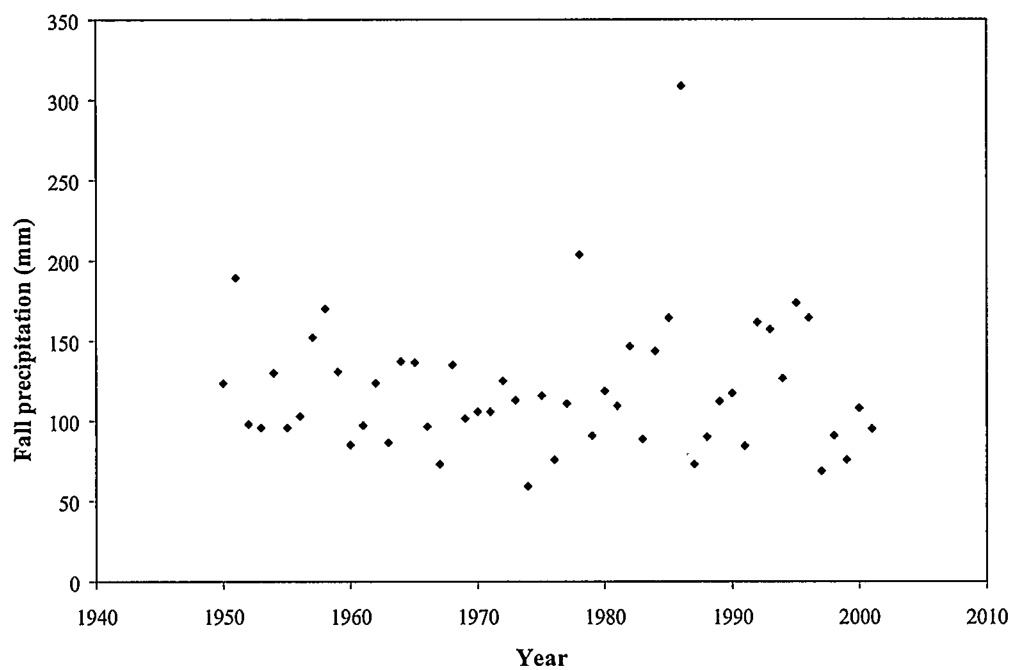

Figure 25. Fall precipitation trends on the Cypress Hills Interprovincial Park – Alberta plateau between 1950 and 2001.

Grazing

Livestock grazing for all stock associations on the CHIP-AB plateau ranged from 14,476 AUM in 1959 to 10,800 AUM in 2005, with a long-term average of 12,277 AUM (Figure 26). Grazing decreased in all three allotments over time. A general trend of increase was observed in AUM-used by elk, with slight decreases in the early 1980s and early 1990s, and a major decline after 2000. Peak elk forage use peaked at about 7,600 cattle equivalent AUM in the late 1990s (Figure 27). Overall livestock and elk AUM-used in the park averaged 18,028 AUM (Figures 28 and 29). Correlation analysis suggests that overall grazing intensity was negatively related to tree establishment (r = -0.580, P <0.001, SEE = 9.463, n = 42) (Figure 28). Forest and isolated tree cover change (ha), as determined from the spatial and temporal analysis (Table 5), was also negatively correlated to overall grazing intensity (r = -0.541, P <0.001, SEE = 156.45, n = 47) (Figure 29).

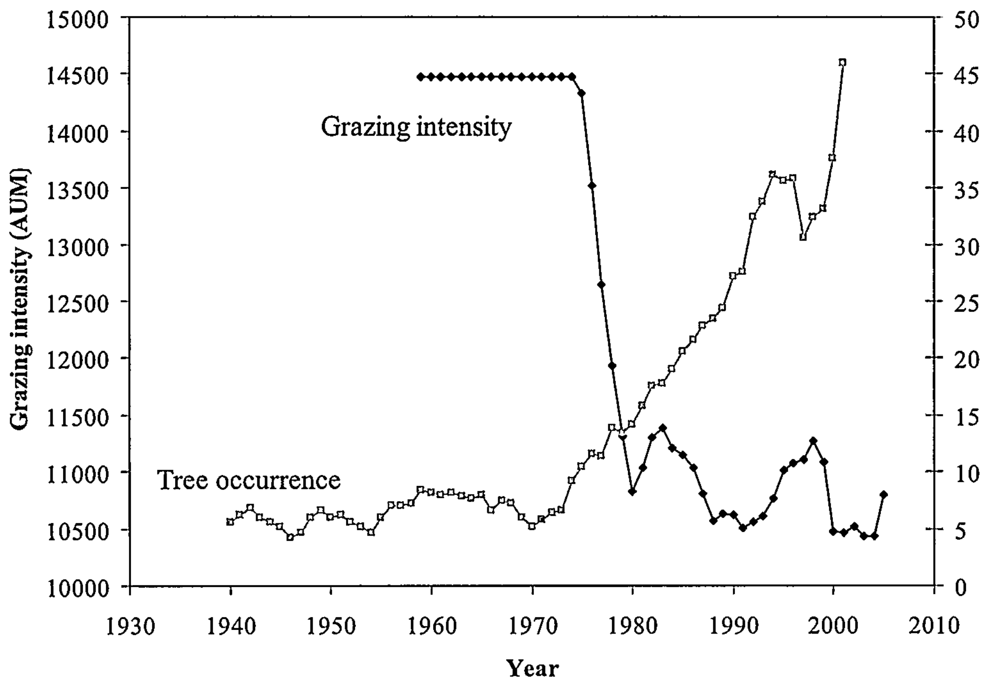

Figure 26. Livestock grazing intensity (1959–2005) and tree establishment numbers (1940–2001) on the Cypress Hills Interprovincial Park-Alberta plateau.

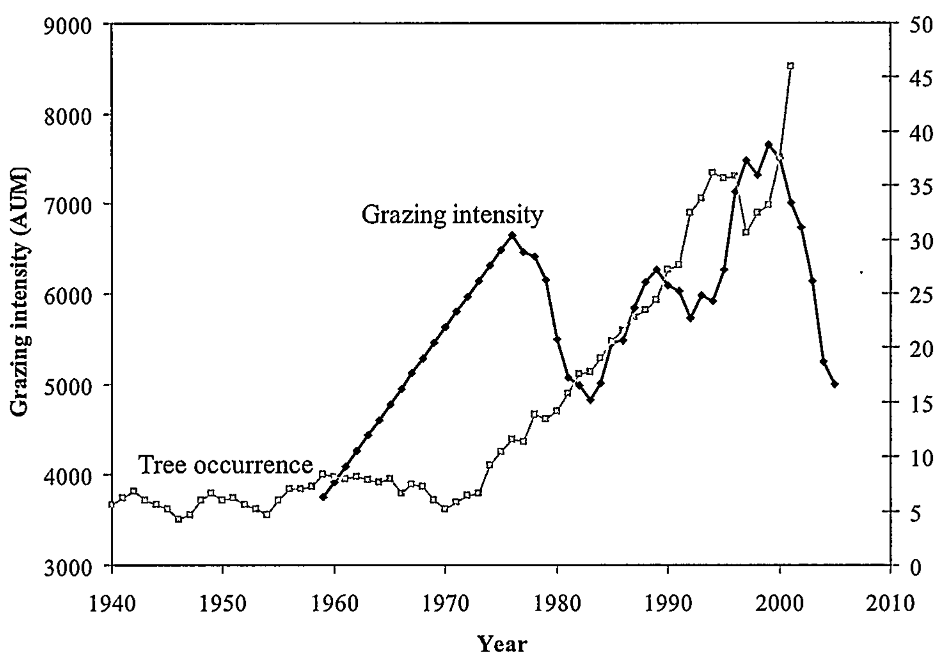

Figure 27. Elk grazing intensity (1959–2005) and tree establishment (1940–2001) on the Cypress Hills Interprovincial Park-Alberta plateau.

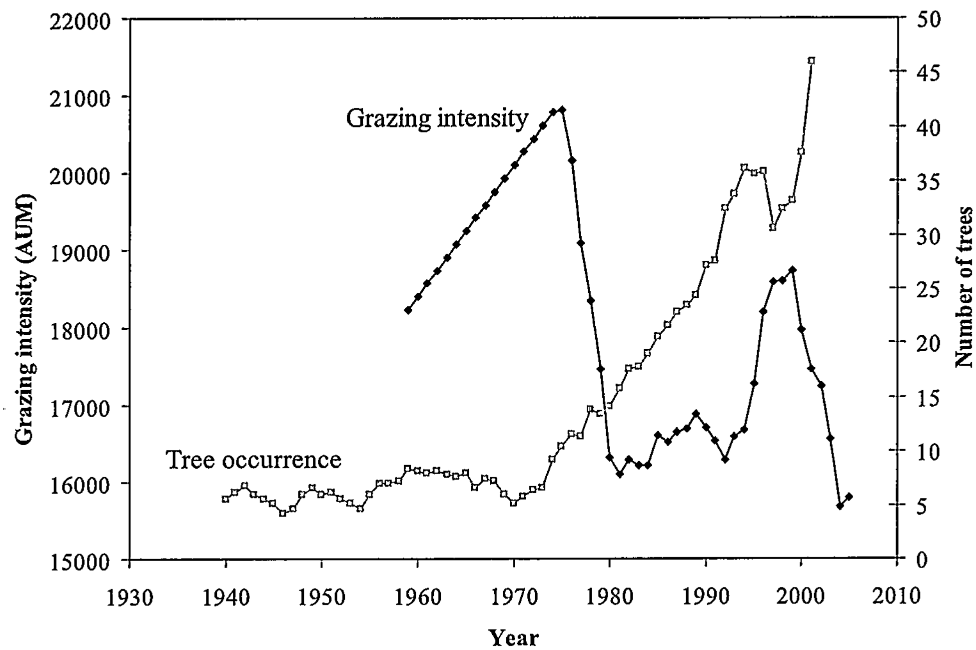

Figure 28. Combined livestock and elk grazing intensity (1959–2005) and tree establishment numbers (1940–2001) on the Cypress Hills Interprovincial Park-Alberta plateau.

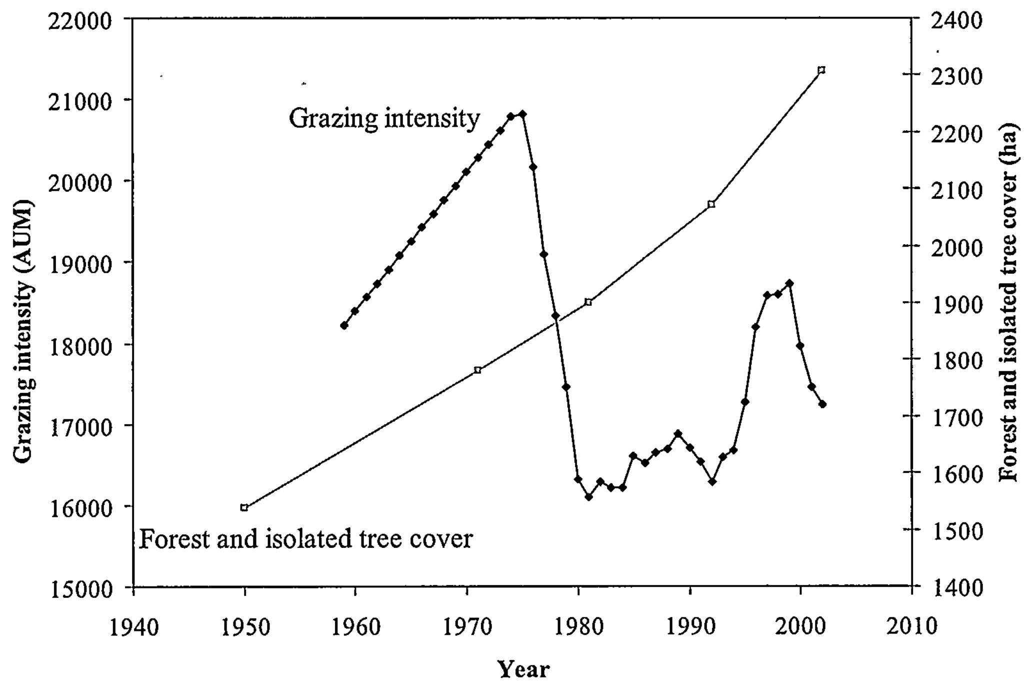

Figure 29. Combined livestock and elk grazing intensity (1959–2005) and amount of new forest and isolated tree cover (1950–2002) on the Cypress Hills Interprovincial Park-Alberta plateau.

CHAPTER 4: DISCUSSION

Forest and trees have encroached on the fescue grasslands of the Cypress Hills Interprovincial Park – Alberta plateau since at least the late-1800s based on the collected tree age data. The rate of tree establishment has increased exponentially since the last stand replacing fires (Figure 4). In 2007, 2,311 ha or about 30% of the Cypress Hills plateau was occupied by forest or had been invaded by trees of various sizes and densities. *Pinus contorta* was the most common tree species associated with this encroachment. *Populus tremuloides* was not substantially contributing to encroachment at this time; reproduction by suckers confined new stems to areas adjacent to the forest edge and many do not survive to maturity. *Picea albertiana* was a much less important encroachment species than *Pinus*, because it typically invaded established *Pinus contorta* and *Populus tremuloides* stands, rather than colonizing grassland areas.

About 14,000 years ago, the Cypress Hills were dominated by boreal forest (Strong and Hills 2005), but had a *Populus tremuloides*, grassland, and shrub complex 7,700 years ago, which changed to predominantly grassland during the hot, dry climate of the late-Hypsithermal (Sauchyn and Sauchyn 1991). When the Hypsithermal ended about 6,000 years ago, a mosaic of grassland and forests of *Pinus contorta* and *Picea albertiana* developed when the climate turned cooler and wetter (Sauchyn and Sauchyn 1991). Considering a mosaic of coniferous forest and grassland has been potentially present on the Cypress Hills for at least 6,000 years, it is surprising that fescue grasslands have remained treeless until the latter half of 20th century, and that forest and trees are now increasing in abundance. Although a continued linear rate of encroachment based on area of forest cover would require 350 years for the plateau to become completely forested (Figure 12), the exponential rate of tree establishment (Figure 4) indicates only an additional ~40 years would be required. The timeframe under continued ecological conditions of the recent past might more realistically be between the two estimates, maybe up to an additional ~100 years. Changes in the Cypress Hills environment during the past century have likely caused the observed changes in vegetation.

Woody plant encroachment occurred across western North America during the past century (Franklin *et al.* 1971; Moore and Huffman 2004; Bekele *et al.* 2006; Haugo and Halpern 2007; Strong *et al.* 2009), and in other parts of the world such as Australia, northern Asia, and central Europe (Wearne and Morgan 2001; Sankey *et al.* 2006a; Anthelme *et al.* 2007). The rates of forest expansion (~0.9% per year) and grassland loss (~0.2% per year) in the Cypress Hills are similar to those reported in Alberta and elsewhere in western North America. In the Porcupine Hills, *Populus tremuloides* forest on north facing slopes expanded by ~31% between 1939 and 1954 (2.1% per year), but did not expand in other areas of these hills (Fredrickson 1975). Over 50% of grassland in the Montane ecoregion of Jasper National Park was lost between 1949 and 1991 (1.3% per year) (Rhemtulla *et al.* 2002). Spencer *et al.* (2009) identified a 25% loss in prairie cover in the Loess Hills of southeastern South Dakota between 1941 and 2000, or ~0.4% per year. Coop and Givnish (2007) calculated an 18% decline in the Valles Caldera grasslands of New Mexico between 1935 and 1996 (~0.3% per year). The grassy balds of the Oregon Coast Range in Washington declined 66% in grassland cover between

1949 and 2000, or 1.3% per year (Zald 2009). These similar rates of change suggest that regional or continental processes (e.g., climate change) are influencing forest encroachment.

Other studies have also identified *Pinus contorta* as an important species encroaching on meadows and grasslands (Helms 1987; Jakubos and Romme 1993; Haugo and Halpern 2007). *Pinus contorta* is a highly invasive species, because it can establish in difficult conditions and colonize disturbed as well as grassland areas (Pfister and Daubenmire 1973; Despain 2001). Young trees grow quickly, allowing them to initially out-compete other tree species for canopy space, such as *Picea albertiana* (Pfister and Daubenmire 1973). *Pinus contorta* are able to reproduce as young as five years old, and good seed crops can occur at intervals of one to three years (Pfister and Daubenmire 1973). Although *Pinus contorta* trees are commonly known to produce serotinous cones that open when subjected to temperatures of 45–60°C or higher (e.g., after fire), non-serotinous cones can also be produced that open without high temperature exposure (Smithers 1961). Most trees in young stands, such as those found on the Cypress Hills plateau, produce non-serotinous cones (Smithers 1961). This fast growing, early reproducing pioneer species was colonizing the fescue grasslands in the absence of substantial competition.

Topography based microsites have been found to influence woody plant encroachment (Wu and Archer 2005). The pattern of forest and tree encroachment on the Cypress Hills plateau suggests that microsites and landscape positions that provide favorable conditions may be responsible for the current forest-grassland development pattern. Those in close proximity to dry valleys could reflect the potential importance of increased moisture availability in terrain-related microsites for the initial establishment of trees that later form forest islands. Microsites have previously been linked with increased moisture availability due to snowdrift deposits (Jakubos and Romme 1993), as well as reduced evaporation rates due to aspect shading. Presumably some suitable microsites also exist within the grassland environment that allow tree establishment.

Daubenmire (1968) recognized three types of vegetation ecotones:

1. An abrupt transition as a result of an abrupt discontinuity in environmental conditions;
2. An abrupt transition as a result of plant interactions; and
3. A zone of gradual blending of vegetation types that reflect a gradual blending of two distinct environmental condition complexes.

Abrupt transitions were present in some locations on the plateau; usually at the intersection of a steep side-slope and the plateau edge. The third type of ecotone, or gradual blending, best described the change in vegetation within areas of encroachment, because trees and forest islands often established in grassland away from the forest edge. Two different patterns of tree invasion can occur with the gradual blending of vegetation types (type 3 ecotone): linear (Moore and Huffman 2004) and "nonlinear", or dispersed pattern of tree establishment. Linear or directional (Jakubos and Romme 1993) invasion occurs where encroaching trees become progressively younger towards the interior of a meadow or grassland, whereas nonlinear invasion occurs where forests increase in density by filling spaces around and between existing trees (e.g., Moore and Huffman 2004; Heyerdahl *et al.* 2006). A nonlinear pattern of invasion was most characteristic of encroachment in the Cypress Hills. In addition to establishing on favourable terrain-related microsites, factors such as grazing intensity, fire occurrence, and historic climatic trends might also explain the observed nonlinear pattern of encroachment. Although competition between grasses and trees can be important where abrupt ecotones occur (Daubenmire 1968; Wilson 1998), the dispersed distribution of trees within the ingress zone suggested that competition was not a limiting factor for tree establishment.

It is probable that fewer fires in CHIP-AB due to human suppression since the late 1800s has allowed the ingress and development of trees within the fescue grassland by not periodically destroying newly established trees. Fire has been considered a likely causal factor in several forest encroachment studies (e.g., Heyerdahl *et al.* 2006; Sankey *et al.* 2006a; Coop and Givnish 2007), unless it was specifically discounted as a possibility. For example, Zier and Baker (2006) speculated that fire suppression in the high elevation *Picea – Abies* (spruce-fir) forests of the San Juan Mountains, Colorado, would not have impacted encroachment due to the infrequency of natural fires. Arno and Gruell (1986) determined fewer fires to be an important factor leading to forest encroachment in the Galena area of southwestern Montana, which has a return fire interval of 26 years. Near Wise River, Montana, the natural fire interval was 37 years prior to fire cessation, which was considered a major contributor to forest expansion in the area (Heyerdahl *et al.* 2006). Climate change was also an important causal factor in both of these studies. Natural fire intervals between 24 and 65 years have been observed for the different watersheds of the Cypress Hills, with an average return interval of 46 years (Strauss 2001). Twice this amount of time has passed since fire suppression began, suggesting that lack of fire could have had some influence over the vegetation development of the Cypress Hills. Tree establishment on the plateau has increased exponentially since 1893 following the last stand replacing fires in the Cypress Hills (Figure 4). The nonlinear rate of tree establishment on the plateau following the onset of fire suppression activities, however, suggests fire was not the only causal factor of encroachment. The major increase in the rate of encroachment after ~1975, at least 65 years after fire suppression activities began (Scace 1972) and 85 years after the last major stand replacing fires in the study area (Newsome and Dix 1968), also indicates that other factors in conjunction with reduced fire frequency promoted increased tree establishment. Coop and Givnish (2007) observed a similar time lag between fire events and increased rate of encroachment of *Pinus ponderosa* in the Valles Caldera of New Mexico, and concluded that other causal factors contributed to tree invasion.

Previous research has reported conflicting impacts of cattle grazing, which has been shown to both promote and inhibit woody plant encroachment (Sankey *et al.* 2006b). Sankey *et al.* (2006a) determined the relationship between grazing and tree establishment to be a complex interaction of several factors including livestock type, grazing intensity, involved woody plant species, and other causal factors such as fire and climate, which makes it difficult to disentangle the specific effects of grazing. This complex relationship is also present in Cypress Hills. For example, increased grazing would presumably promote *Pinus contorta* establishment, because its seedlings are shade-intolerant (Pfister and Daubenmire 1973), and decreased vegetation ground cover would increase the amount of sunlight reaching the seedling. In contrast, cattle have been shown to browse directly on, as well as trample, *Pinus contorta* seedlings, and damage increases with increased stocking rate and grazing pressure (Pitt *et al.* 1998). Decreased elk grazing has also been found to facilitate increased tree establishment (White *et al.* 2007). The negative correlation between tree establishment and grazing intensity in the Cypress Hills suggests that decreased grazing could have facilitated increased tree encroachment on fescue grasslands. However, an increased rate of tree establishment was observed before grazing intensity dropped below AUM of previous years in 1979, and the rate of tree establishment continued to increase considerably during a period of relatively constant grazing intensity between 1981 and 1994 (Figure 26). Furthermore, *Pinus contorta* make up a very small portion of cattle and elk diet (0.3 and 0.6%, respectively) in the Park (Lee 1979). It is therefore unlikely that changes in grazing intensity substantially impacted forest encroachment, although the complexity of the interaction makes it difficult to accurately determine the nature of the relationship in the Cypress Hills. It is possible that deer have had an impact on the regeneration of seedlings, because *Pinus contorta* make up a much greater portion of their diet than either cattle or elk at 13.1% (25.4% of their winter diet) (Lee 1979). Unfortunately, deer population data were not available for comparison against tree establishment numbers. It is plausible that the deer population has increased since the early 1900's, when hunting was prevalent in the area (Soper 1946). Management hunts in the Cypress Hills only target elk, and the sole predator of deer in the area, cougars, only recently inhabited the Park.

Climatic warming has been recognized to have occurred in Canada during the 20th century (Zhang *et al.* 2000). The average annual temperature in the Cypress Hills increased ~0.6°C between 1929 and 2005, which was within the range of warming observed for southern Canada (below 60°N latitude) between 1908 and 1998 (Zhang *et al.* 2000). The average annual temperature for the Canadian prairies increased by 1°C during the 20th century, with changes in winter and spring accounting for much of the warming (Cutforth *et al.* 2004); findings that were consistent with those of this study. During the latter half of the 20th century, the length of the frost-free season increased by an average of 3.1 days per decade across the same area (Cutforth *et al.* 2004). Warming trends were greatest in the northern agricultural area of Alberta and the Peace River region (Berry 1991; Cutforth *et al.* 2004), where temperatures increased 0.35°C per decade between 1908 and 2004 (Strong *et al.* 2009). The rate of change in the length of the frost-free season in the Cypress Hills between 1940 and 1997 was three to six times that of the surrounding prairies of southern Alberta and Saskatchewan (Cutforth *et al.* 2004). A trend of increasing rainfall amounts and frequency has also been observed in the Canadian prairies since the late 1960s (Gan 1998; Cutforth 2000), with precipitation increasing in the Cypress Hills between 1929 and 2005 (Figure 17). The Cypress Hills are considered a "unique biological-climatological environment" compared to the surrounding prairies, demonstrating an "oasis effect" characterized by a cooler, moister climate (Holmes 1969), which apparently renders them more susceptible to climatic change.

Establishment numbers of all three tree species appeared to be related to certain climatic variables. Most important of these variables was spring temperatures for *Pinus contorta* establishment. The steady increase in spring temperature in the Park has resulted in longer frost-free seasons, which allow earlier phenological development and a longer growing-season (Cutforth *et al.* 2004). In a study of global biological trends in response to climate warming, Parmesan and Yohe (2003) found a significant mean advancement of spring by 2.3 days per decade. The correlation of fall precipitation to establishment numbers reflects the need for adequate soil moisture in early spring to initiate seed germination, since early precipitation is often not sufficient to allow germination (Murphy 1970). Moisture deficiency is a leading cause of mortality of *Pinus contorta* seedlings (Despain 2001; Chhin and Wang 2002), and Arno and Gruell (1986) found accelerated invasion of *Pinus contorta* occurred concurrently with a period of unusually moist conditions in southwestern Montana. The increased moisture availability in the Cypress Hills due to additional precipitation and cooler summer temperatures has likely resulted in additional tree establishment on the plateau; increased moisture has previously been linked to forest development in grassland areas in northern Alberta (Strong *et al.* 2009). The sharp increase in tree establishment after 1975 coincided with a period of greater precipitation, warmer spring temperatures, and cooler summer temperatures than previously recorded. A longer frost-free season, combined with greater moisture availability, likely created an environment conducive to the establishment and development of coniferous trees and forests on the Cypress Hills plateau.

In conclusion, a decreased fire frequency due to active suppression, combined with increased moisture availability and a longer frost-free season, appear to have contributed to the encroachment of forest and trees on fescue grasslands on the Cypress Hills plateau. The absence of fire allowed encroachment to continue unabated in a climatic environment suitable for forest and tree development. If the favourable climatic conditions continue, the rate of tree establishment on the plateau will likely continue to increase. In this environment, natural fires would likely be less frequent due to increased moisture availability. In contrast, global warming predictions indicate that temperatures could continue to rise considerably, which could lead to decreased moisture availability because precipitation is not expected to increase at the same rate (Hengeveld 2000; Houghton *et al.* 2001). If these latter predictions are accurate, the incidence of fires will likely be more frequent even if a suppression regime is continued. The warmer and drier climate conditions would favour fescue grasslands on the plateau rather than trees and forest, and encroachment would no longer be a concern for the Park.

CHAPTER 5: MANAGEMENT OPTIONS AND RECOMMENDATIONS

Management options

Conservation of the extent, health, and native biodiversity of the fescue grasslands is a management objective of Cypress Hills Interprovincial Park–Alberta. Ecological reasons for lessening the loss of fescue grasslands to forest encroachment include its rapid decline in extent across western North America and the associated loss of biodiversity. Fescue grassland is more vigorous, has a higher species richness and diversity than the surrounding prairies, and provides higher quality forage for native and domesticated grazers. Forest – grassland transitions are known to be productive habitat due to the forage provided by grassland, and the shelter provided by forest and trees (Shugart 1998). If grassland is completely lost, the abundant population of ungulates in the park may no longer be supported, which would threaten the food supply of the cougar, which has recently colonized the park. Sprague's Pipit may be lost from the Park, as it relies on fescue grasslands for habitat. Hull (2002) speculated that a mosaic of forest, trees, and grassland created ideal habitat for birds on the plateau, but additional research was needed to identify the effects of forest encroachment on bird diversity and habitat. Therefore, the following vegetation management options should be considered within the context of an overall resource management plan, if the objective is to maximize the extent of fescue grassland within the Park (Table 11).

Table 11. Potential vegetation management options to reduce tree abundance on the Cypress Hills Interprovincial Park-Alberta plateau, and associated outcomes, benefits, and concerns.

Management Options	Anticipated outcome for plateau vegetation	Benefits	Concerns
1). Continued fire suppression	*Pinus contorta* forest continues to increase in areal extent, and grassland abundance decreases	• Minimal management intervention (aside from fire suppression) • No additional associated costs	• Potential loss of valued ecological resources (e.g., Sprague's pipit, grazing forage) • Build up of woody biomass with greater fire hazard
2). Thinning (selective cutting) of mature trees	Reduced abundance of *Pinus contorta* trees and increased fescue grassland coverage	• Potential for immediate changes to the vegetation distribution • Potential for economic gain in the case of commercial logging • No risk to people and infrastructure	• Public acceptance
3). Manual removal of young trees	Reduced abundance of young trees and increased fescue grassland coverage	• Prevent additional or slow encroachment • No risk to people and infrastructure	• Public acceptance • Increased danger of fire due to fuel build up, or removal of nutrients from system • Repeated removal of seedlings would be required as they become established • Costs incurred due to associated labour
4). Mowing	Reduced abundance of young trees	• Low cost • No risk to people and infrastructure	• Effectiveness of method is unknown • Can only remove seedlings and saplings above the mowing height, seedlings and mature tree coverage • Would not necessarily kill trees

Table 11. Concluded.

Management Options	Anticipated outcome for plateau vegetation	Benefits	Concerns
5). Chemical treatment of conifers with herbicide	Reduced abundance of young trees	• Low cost • No risk to people and infrastructure	• Increased fire hazard due to woody debris, or removal of nutrients from system • Public acceptance • Long term effects on vegetation and wildlife not well understood
6). Prescribed fire program	Mosaic of fescue grassland, *Pinus contorta* and *Populus tremuloides* forest patches	• Mimics a natural disturbance agent • Potential for immediate vegetation change • Increase in quantity/quality of forage for grazing • Increases in biodiversity • Reduce danger of high intensity fire due to reduction in woody biomass and small shrub vegetation	• Danger associated with fire due to close proximity of Elkwater (risks to people and infrastructure) • High cost of implementing program • Public acceptance

Management Option 1: Continued fire suppression

Continued fire suppression with no additional management intervention would result in continual *Pinus contorta* forest and tree development on the plateau. In as little as 40–100 years, all fescue grassland could be lost to *Pinus contorta* forest. This option is a preference and recommendation of some stakeholders, because with the exception of an altered disturbance regime due to the suppression of wildfires, it is considered an ecologically-based approach (Western Ecological Services Ltd. 1991; Locke 1992). No additional expense is incurred with continuation of this management option. The additional fuel load resulting from fire suppression would greatly increase the risk of future high intensity fire occurrences, however, and the loss of important ecological resources of the Park in the long-term may not be worth the short-term monetary benefits. Fescue grasslands are an important resource, and their loss will lead to a decline in forage, habitat, and biodiversity in the Park as well as at a national-level.

Management option 2 and 3: Thinning/selective cutting

Thinning and selective cutting would involve the removal of selected trees or groups of trees from the fescue grasslands by mechanical or manual means; mechanical methods would be required for the removal of larger trees of possible commercial value, and manual methods could be used for removal of smaller trees. This option has been discussed as a method of shrub removal to slow or halt encroachment in savannahs (Smit 2004; Brudvig and Asbjornsen 2008, 2009). Its effectiveness as a method of managing forest and tree encroachment is not adequately documented, although it has been used for this purpose in protected areas in combination with prescribed burning (Stevens *et al.* 2004). This approach would result in an immediate change to the vegetation in favour of non-woody plants (Smit 2004). However, the current rate of tree establishment will continue or increase unless seed sources are substantially reduced. A considerable number of trees (e.g., maybe >100,000) would need to be removed to sufficiently deplete the sources of *Pinus contorta* seeds.

The benefits of selective cutting include potential for economic gain through logging, but this may not be consistent with Park objectives, and a major concern associated with this option is that the public perception may be negative. The Forest Management Workshop in 1991 in Medicine Hat, Alberta (Locke 1992), was held in response to growing public concern following the harvesting of timber in the late 1980s. Potential damage to the ground surface and surrounding vegetation incurred from equipment should also be considered; however, removal during winter using light equipment would minimize physical damage to the ground surface and surrounding vegetation. The killing of small trees has the potential to increase fire hazard due to build up of fuels, but if cut trees are removed from the cleared areas, costs will be incurred due to the labour required and disposal.

<u>Management option 4: Mowing fescue grassland</u>

Rangeland management practices such as mowing and haying have not been adequately investigated to understand their effectiveness in lessening the loss of grasslands to forest and tree encroachment. Mowing can produce effects similar to grazing, including increased vigour, abundance and primary production of the grassland community (Collins *et al.* 1998). Mowing has been shown to reduce shrub coverage in grassland (Van Dyke *et al.* 2004; Zuckerberg and Vickery 2006), and could possibly be an effective method of removing seedlings and saplings above the mowing height (e.g., 30 cm, Van Dyke *et al.* 2004). If only the tops of trees are cut, the tree could respond with more vigorous growth and multiple upright growing branches. Haying, as an alternative, causes a loss of nutrients from the system; fertilizer additions are often used to replenish lost nutrients from haying in pasture (Sigua *et al.* 2006). The latter would represent a major interference with the functioning of the natural ecosystem of the area and is not recommended.

Management option 5: Chemical treatment of conifers

The chemical treatment of conifers would involve the application of herbicides to kill small trees and seedlings, and could be used in combination with mechanical removal of larger trees. The herbicide Oracle (Gharda USA Inc. 2004) is used for the removal of coniferous saplings and seedlings from rangeland, including red fescue (*Festuca rubra*) areas. An important concern with the application of a herbicide is its impact on the surrounding, non-targeted vegetation and wildlife. The chemical effects of herbicides on vegetation have been found to last more than 17 years and long-term effects are unknown (Strong and Sidhu 2005). Spot treatment as opposed to general application would help to reduce the affects on nearby vegetation and wildlife, but is more labour intensive. Public acceptance for this option will likely be low, since it directly interferes with the natural ecological process, and involves the addition of chemicals to the system. This option is not recommended due to probable low public acceptance, poor understanding of the effects on local vegetation and wildlife, and its considerable interference with natural ecological processes.

Management option 6: Prescribed burn program

A prescribed burn program could be introduced as a means of eliminating existing tree encroachment and minimizing future tree establishment. Such programs have effectively reduced woody plant encroachment in other parks and protected areas (Stevens *et al.* 2004; Ducherer *et al.* 2009). Following prescribed fires, vegetation on the plateau will likely be composed largely of fescue grasslands, intermixed with forest and tree patches. The fire interval for the Cypress Hills has been identified as between 24 and 65 years, comprised of frequent, low intensity fires, and infrequent, high intensity fires (Strauss 2001). The species composition of forest and trees on the plateau would depend on the intensity of the fires. Ducherer *et al.* (2009) reported that low intensity burning was effective for removing *Pinus ponderosa* (ponderosa pine) saplings and seedlings, and increasing grassland and forb biomass; whereas crown fires of a few hundred hectares are required to remove spruce seed sources (Despain 1983). A large, high intensity fire has greater potential to get out of control, and therefore, is a greater danger to people and infrastructure. Seasonality of prescribed burning must also be considered. Burning is least damaging to grasses when dormant (Wright 1974), and fires during the fall season have a greater negative impact on grasses, especially *Festuca* spp., than spring fires

(Redmann *et al.* 1993). Fall fires also reduce winter shelter and forage for wildlife such as elk (White 2001).

A prescribed burn program would meet the park objective of conserving fescue grassland, while minimizing human intervention by mimicking a natural disturbance agent. The maintenance of the fescue grassland community would help to conserve habitat and biodiversity on the plateau. An increase in the quantity and quality of forage for native and domestic grazers would likely result. Furthermore, prescribed burns could reduce the potential for mountain beetle infestation (Parks Canada 2009a). Prescribed burns should not increase the establishment of *Pinus contorta* from serotinous cones, because the majority of this species in grassland areas are young and likely produce non-serotinous cones. The most important concern associated with the implementation of a prescribed burn program is the potential danger to people in proximity to the park, as well as the potential loss of wildlife habitat and damage to infrastructure. However, the success of prescribed burn programs implemented by the Parks Canada Agency in national parks such as Banff shows that these concerns can be addressed (Low *et al.* 2009). The relatively small size of Cypress Hills Interprovincial Park in comparison to other parks with prescribed burn programs may make the short-term damage conspicuous and potentially less acceptable to the public, and this should be considered. Education programs, such as those currently implemented by the Parks Canada Agency (2009b), enhance public acceptance. Monetary costs of a prescribed burn program should also be considered; the total cost of burning areas of ingress on the Alberta section of the Cypress

Hills plateau would be approximately $60,000 at an average cost of $80 per hectare (Parks Canada Agency 2009b).

Recommendations

If fescue grasslands are to be maintained in the park, a combination of methods may be necessary to achieve the desired outcome of less tree encroachment and increased extent of grassland vegetation. Initially, it is recommended that a prescribed burn program be implemented. This should involve experimentation with low intensity fires on small portions of the park in early spring, and monitoring of the impacts on the vegetation composition, as well as on wildlife populations and their habitat. Such experimentation was also conducted in Banff National Park over a period of ~10 years prior to the implementation of their fire management program (Low *et al.* 2009). The desired mosaic of fescue grassland and forest may be attainable without the need for high intensity fires. However, if lower intensity fires are not sufficient to stop or slow the invasion of trees and the development of new forest, more higher intensity may needed.

Alternatively, low intensity fires focused primarily on the elimination of young trees in grassland dominated areas could be combined with the mechanical removal of larger trees and forest islands in grassland. In addition to increasing fescue grassland vegetation, the removal of larger trees would reduce forest canopy closure and lessen the build-up of fuels, effectively decreasing the likelihood of high intensity fires. Once encroachment is under control, occasional burning (e.g., every 10 to 15 years) of the resultant grassland would remove subsequently established seedlings and saplings. A public education program, such as "Bighorn in your Backyard" implemented by the Parks Canada Agency for Kootenay National Park (Dubois *et al.* 2004), would help to improve public awareness and acceptance of forest thinning and prescribed burns. Conservation of the fescue grasslands on the Cypress Hills Interprovincial Park – Alberta plateau should be possible through an ecologically-based approach to management that mimics fire as a natural disturbance agent, includes monitoring of the resultant impacts on vegetation and wildlife, and involves public education regarding management objectives, actions and anticipated outcomes.

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APPENDIX A: TREE-AGE ESTIMATION

For *Populus tremuloides*, a linear regression model with a forced *y*-intercept of 0 was used to estimate age of incomplete and rotten cores from basal diameter, with r = 0.92 and P <0.001 (Figure A1). A linear regression model was also used to estimate ages of *Pinus contorta*; this model did not have a *y*-intercept of 0, however only trees with a basal diameter >10 cm required aging (r = 0.67, P <0.001) (Figure A2). A logistic regression model was used to estimate ages of *Picea glauca,* with R = 0.77 and P <0.001 (Figure A3).

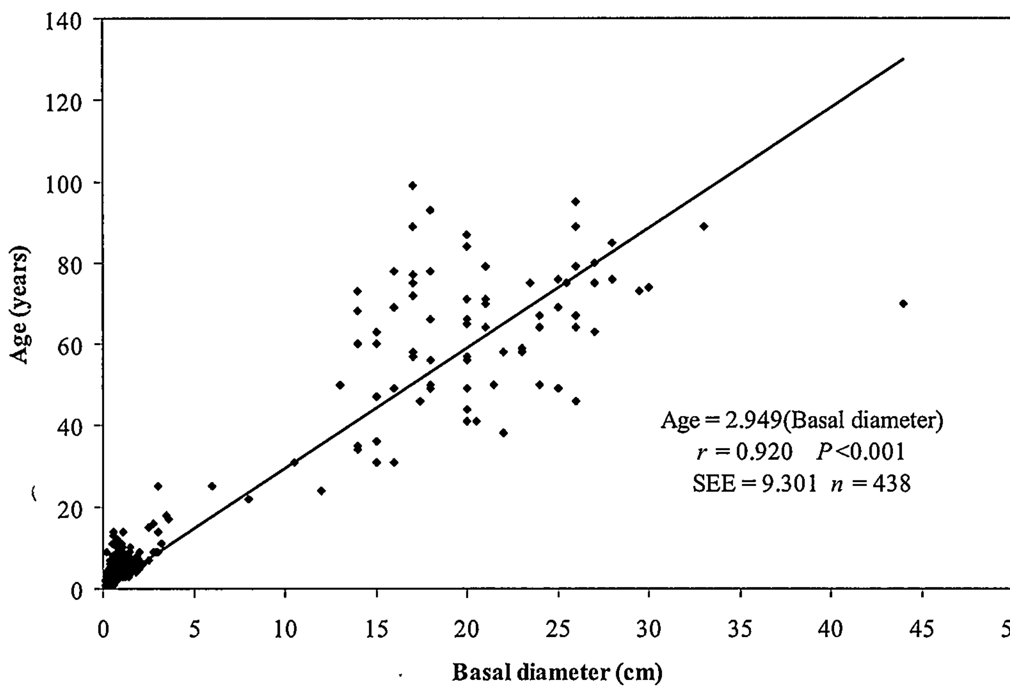

Figure A1. Basal diameter – age curve of *Populus tremuloides* used to estimate ages of trees with incomplete or rotten increment cores.

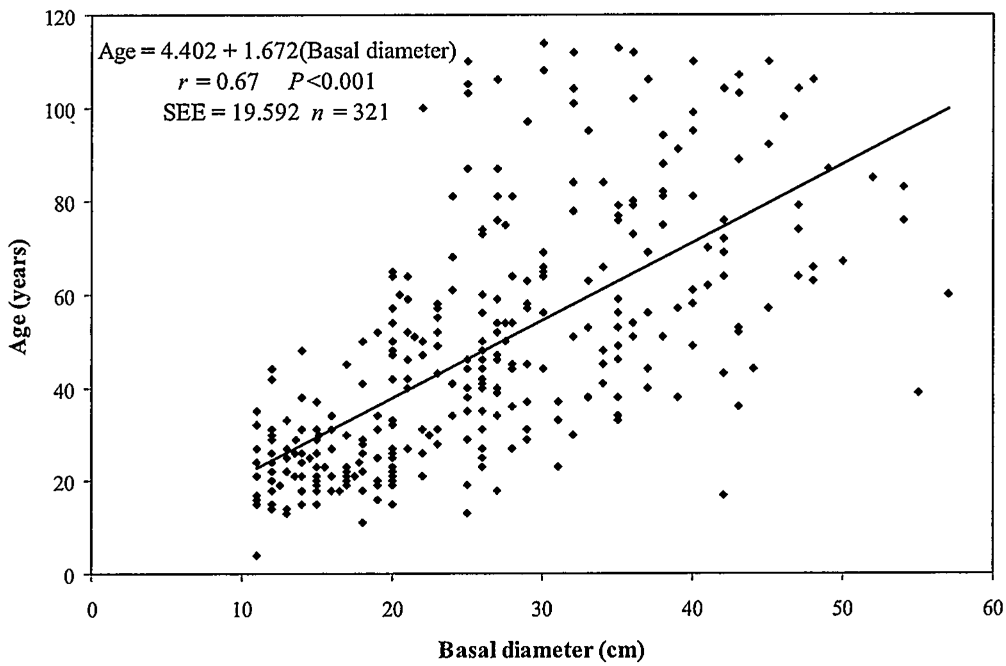

Figure A2. Basal diameter – age curve of *Pinus contorta* used to estimate ages of trees with incomplete or rotten increment cores.

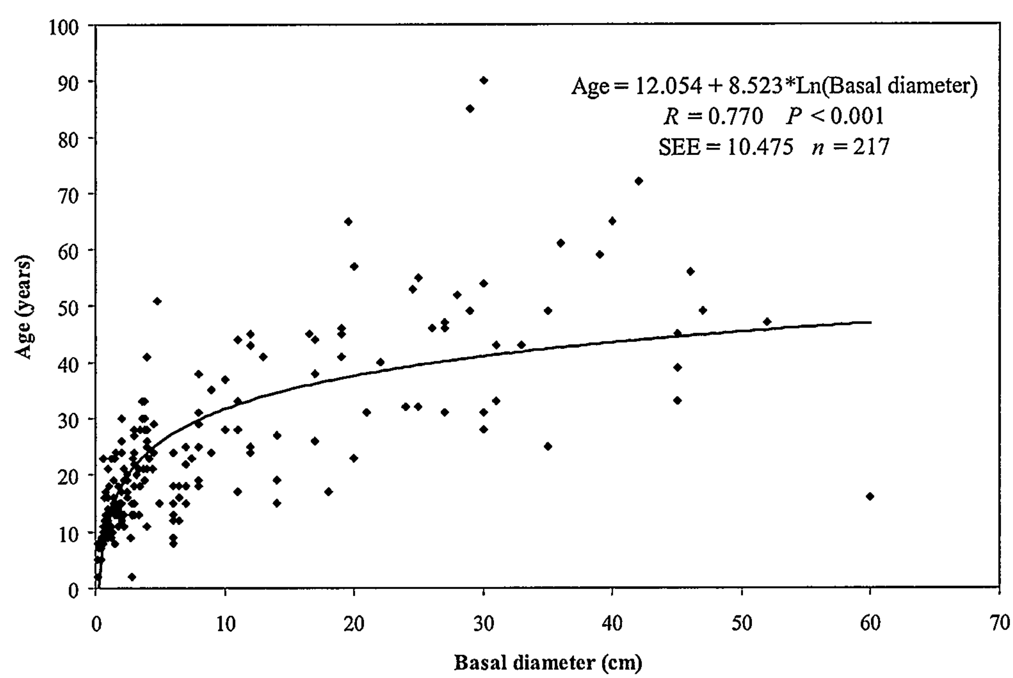

Figure A3. Basal diameter – age curve of *Picea albertiana* used to estimate ages of trees with incomplete or rotten increment cores.

APPENDIX B: Degree of fit between Medicine Hat and Cypress Hills climate stations

The following diagrams illustrate the degree of numerical fit between the Medicine Hat Airport and Cypress Hills stations, as well as the equations used to estimate temperature and precipitation records for the CHIP – Alberta plateau. See Methods for details related to data adjustments.

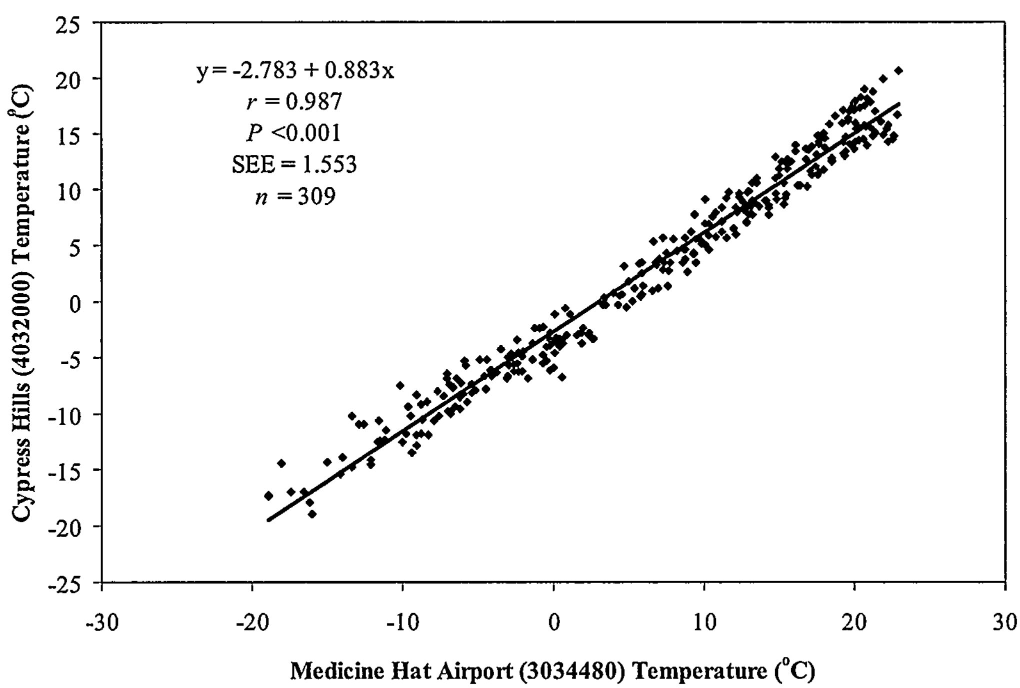

Figure B1. Cypress Hills Park (4032000) and Medicine Hat Airport (3034480) temperature records.

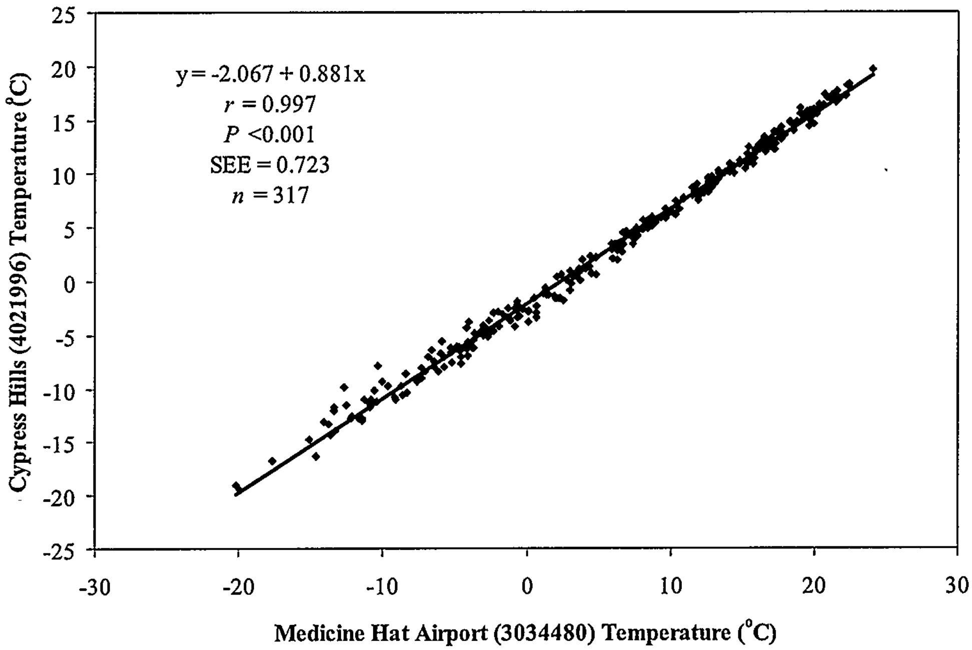

Figure B2. Cypress Hills (4021996) and Medicine Hat Airport (3034480) temperature records.

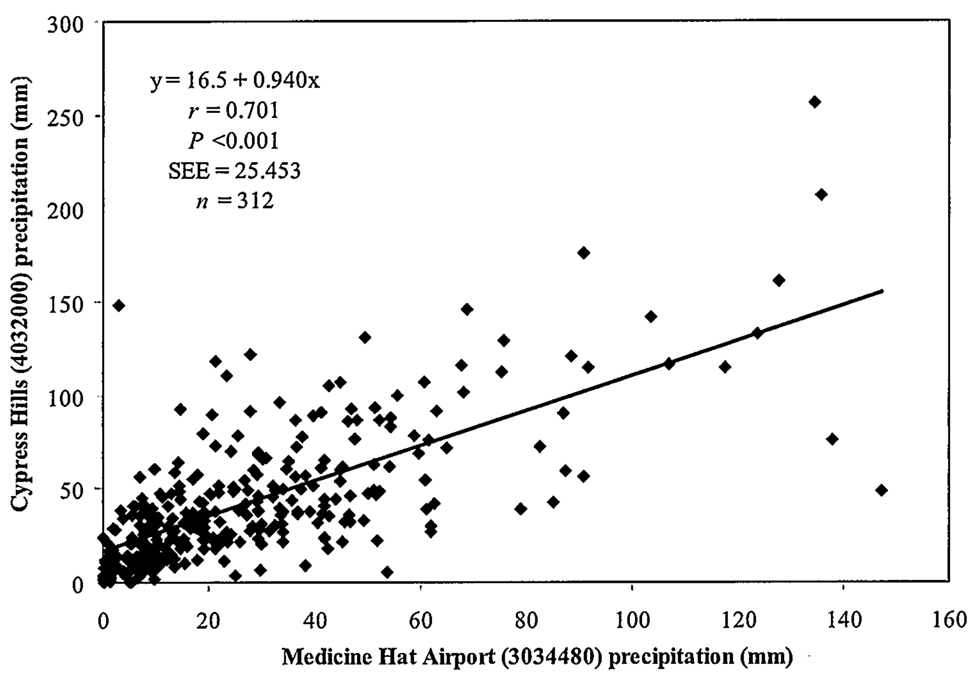

Figure B3. Cypress Hills Park (4032000) and Medicine Hat Airport (3034480) precipitation records.

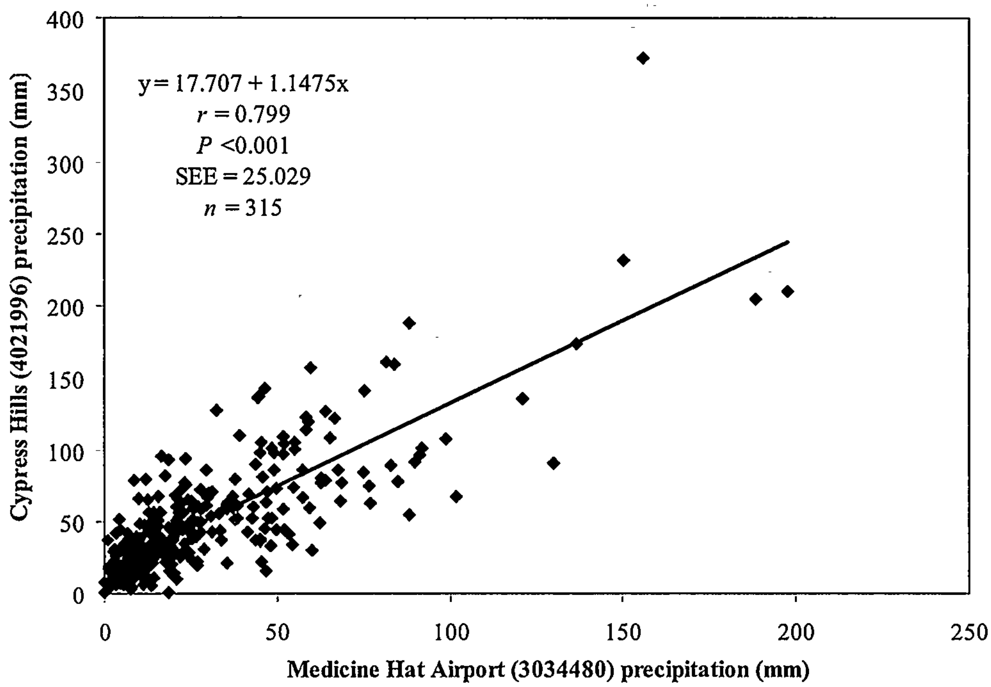

Figure B4. Cypress Hills (4021996) and Medicine Hat Airport (3034480) precipitation records.

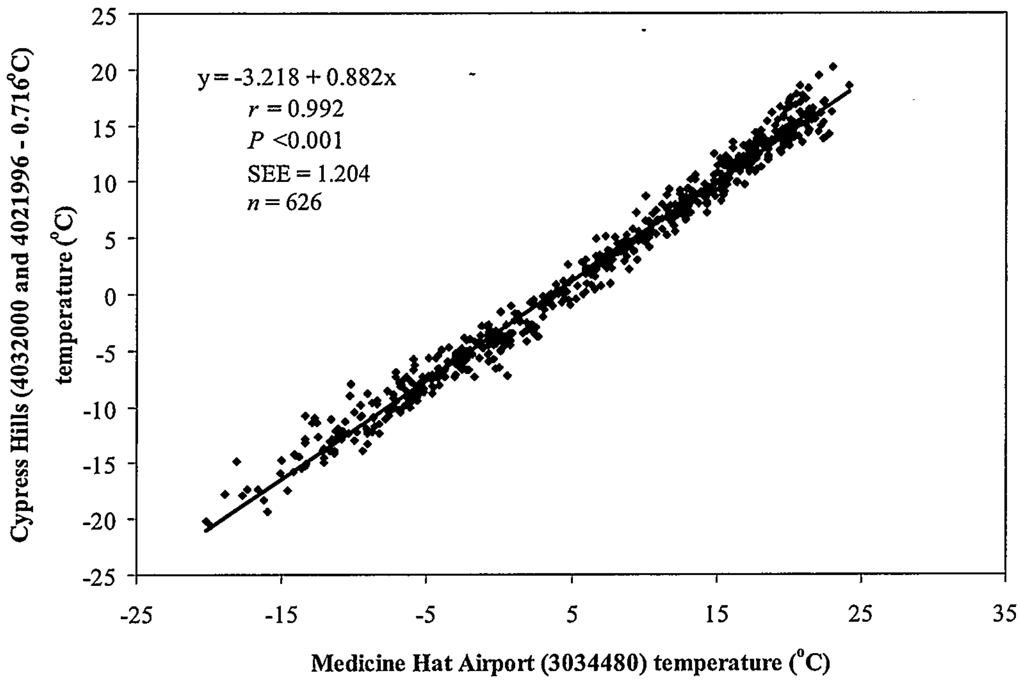

Figure B5. Relationship between Cypress Hills (4032000 and adjusted 4021996), with correction (0.716°C) for difference in elevations with respect to plateau surface, and Medicine Hat Airport (3034480) temperature data used to estimate study area temperatures.

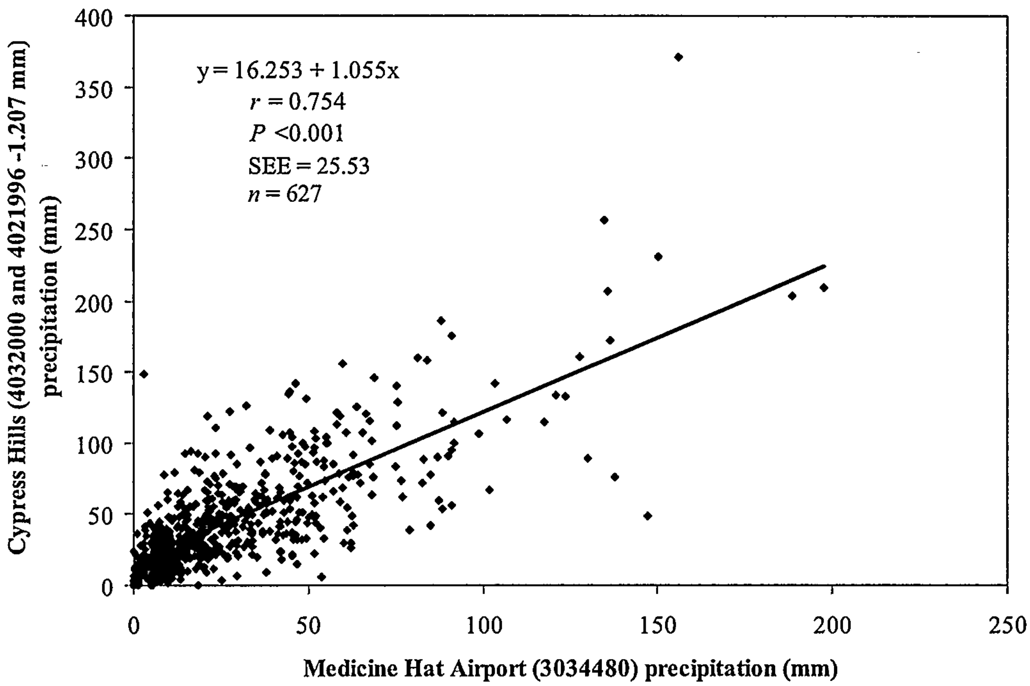

Figure B6. Relationship between Cypress Hills (4032000 and adjusted 4021996) and Medicine Hat Airport (3034480) precipitation data used to estimate study area precipitation.

APPENDIX C: Forest Cover Maps

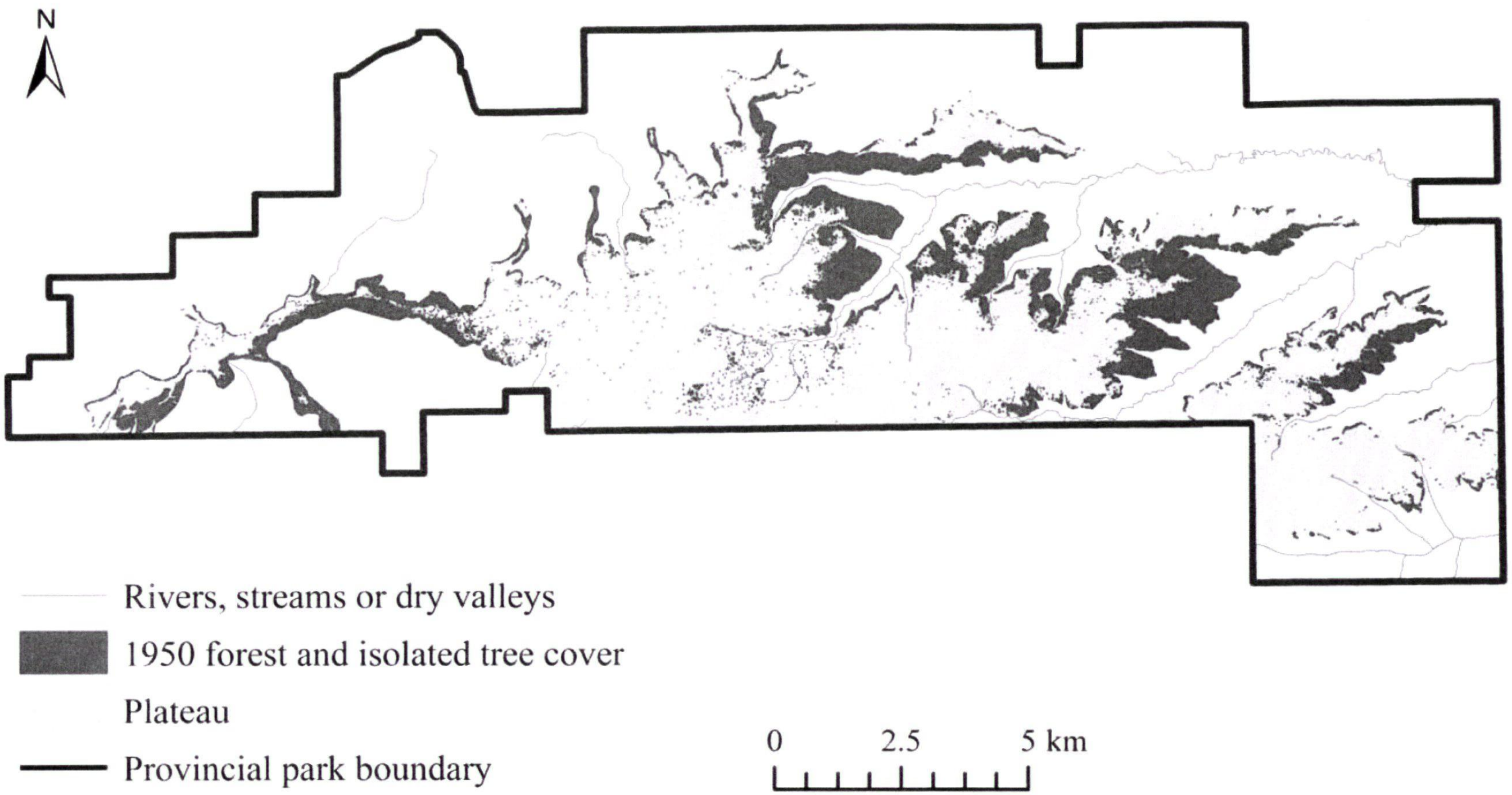

Figure C1. Forest and isolated tree cover of the Cypress Hills Interprovincial Park – Alberta plateau in 1950.

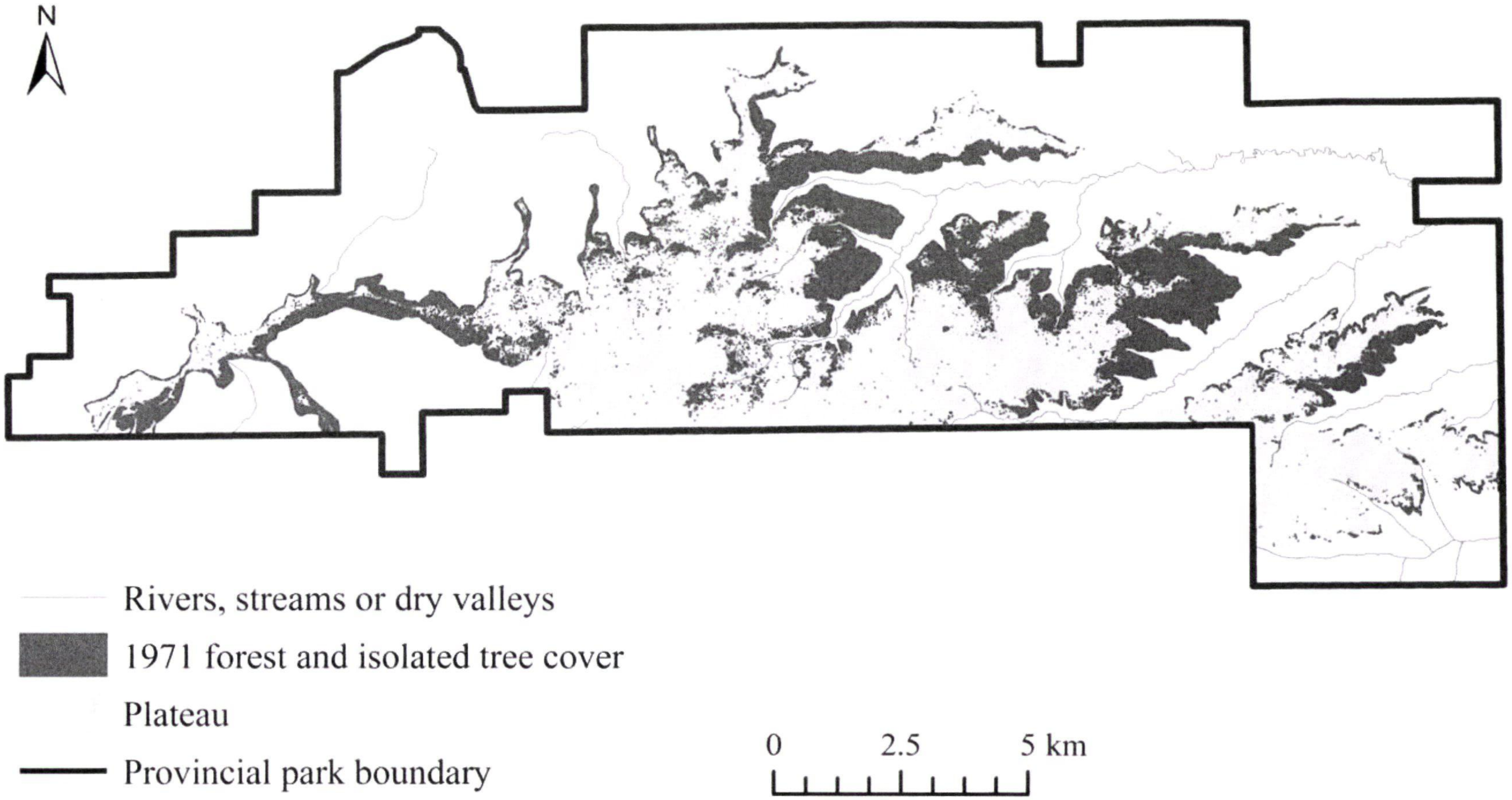

Figure C2. Forest and isolated tree cover of the Cypress Hills Interprovincial Park – Alberta plateau in 1971.

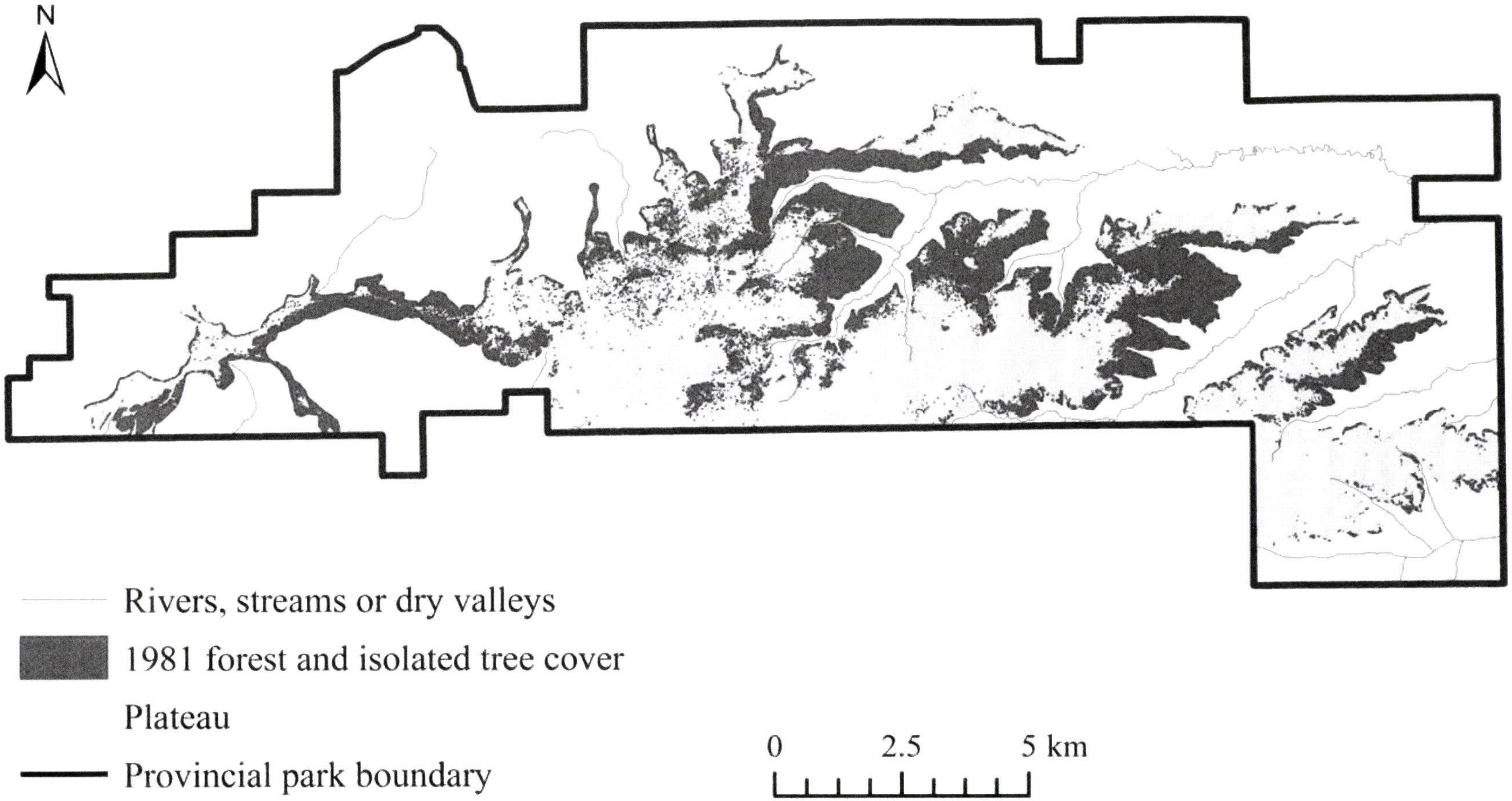

Figure C3. Forest and isolated tree cover of the Cypress Hills Interprovincial Park – Alberta plateau in 1981.

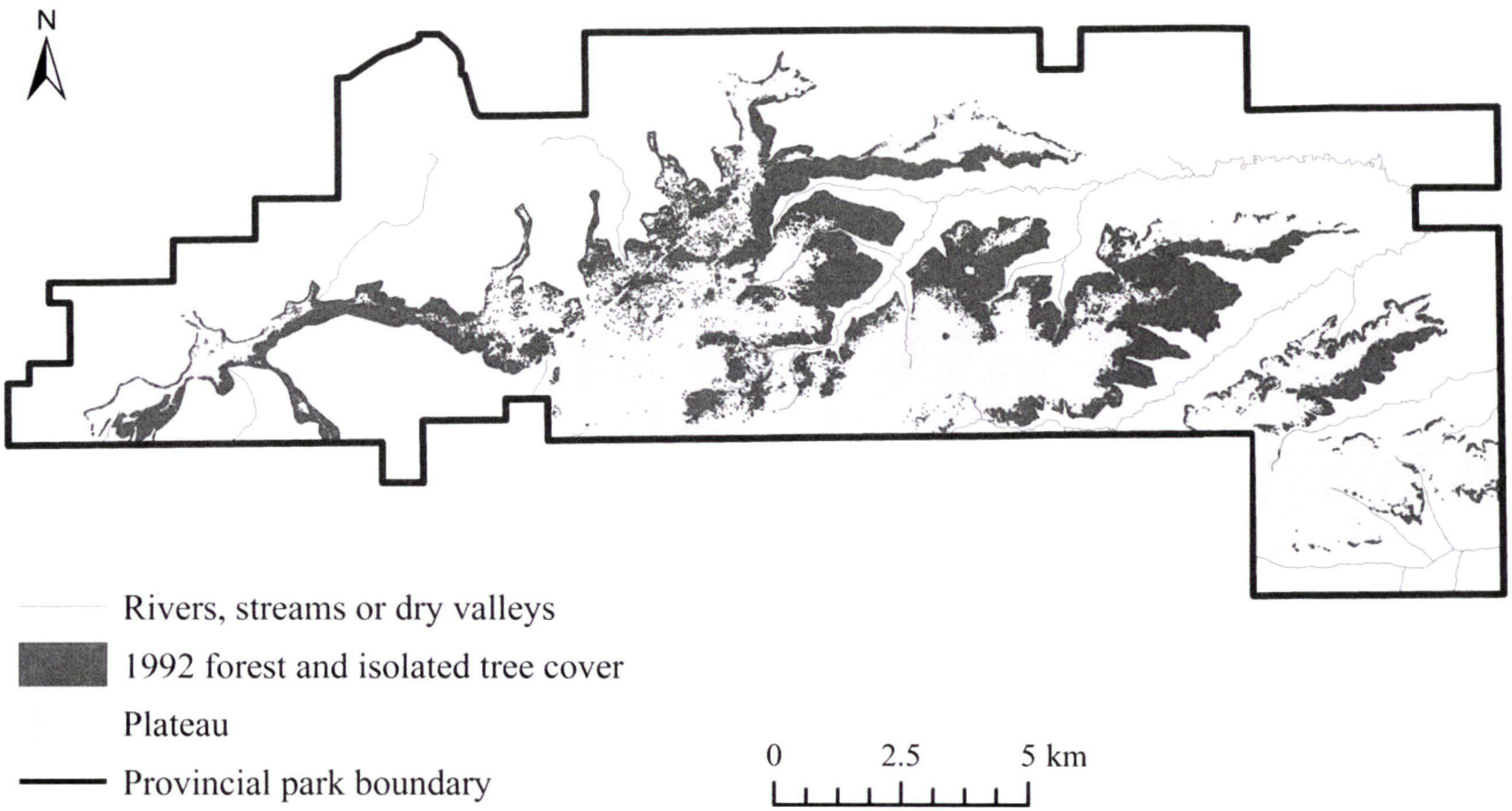

Figure C4. Forest and isolated tree cover of the Cypress Hills Interprovincial Park – Alberta plateau in 1992.

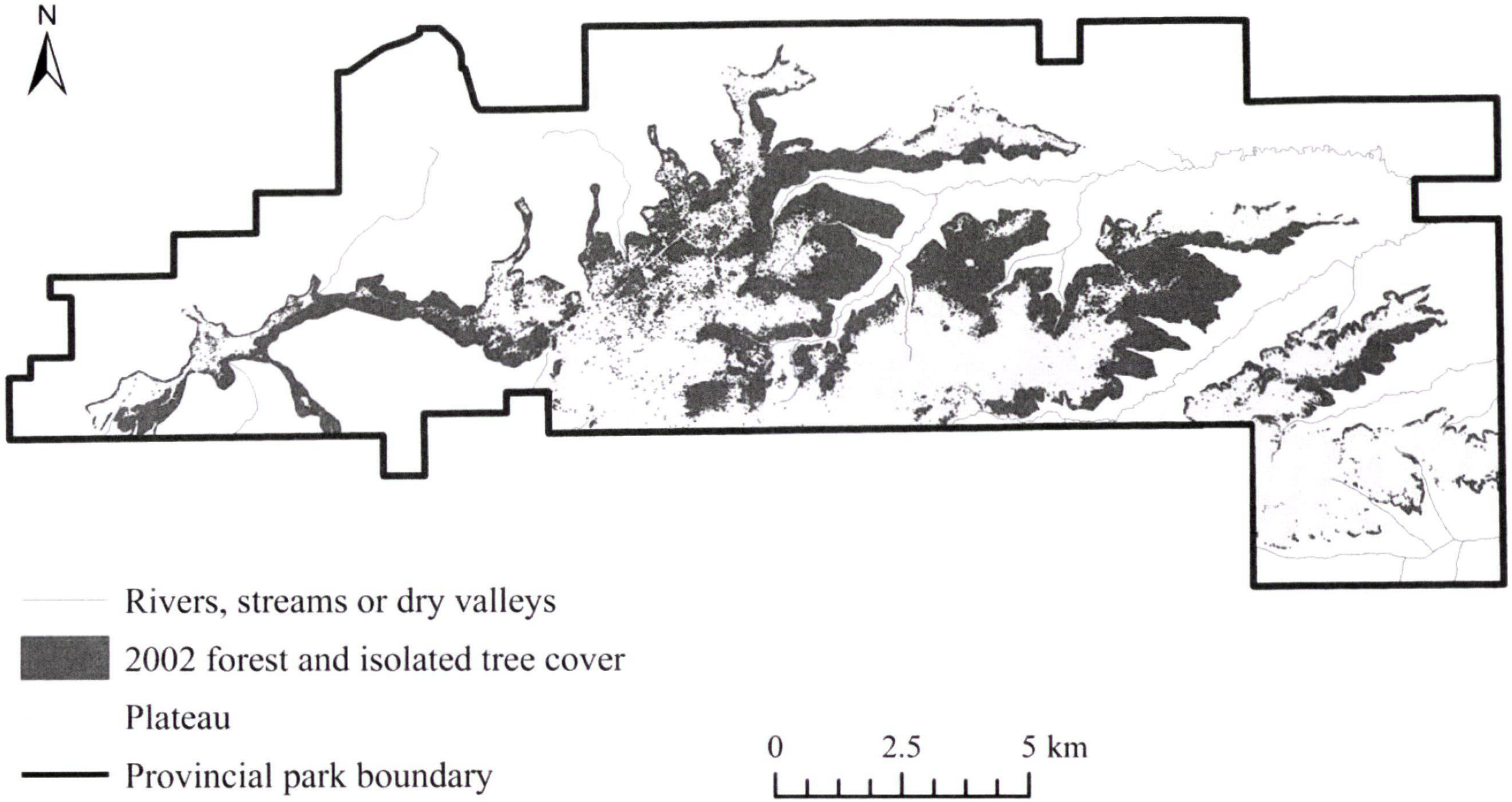

Figure C5. Forest and isolated tree cover of the Cypress Hills Interprovincial Park – Alberta plateau in 2002.