Fire regimes of the Alaskan boreal forest

James Cronan, Donald McKenzie, and Diana Olson

James Cronan is a research scientist, **Donald McKenzie** is a research ecologist, and **Diana Olson** is a University of Idaho-FRAMES project manager, U.S. Department of Agriculture, Forest Service, Pacific Wildland Fire Sciences Lab, 400 N 34th Street, Suite 201, Seattle, WA 98103.

Fire regimes of the Alaskan boreal forest

James Cronan, Donald McKenzie, Diana Olson

Authors

U.S. Department of Agriculture, Forest Service
Pacific Northwest Research Station
Portland, OR
General Technical Report PNW-GTR-XXX
Month 201x

Published in cooperation with:
University of Washington, School of Forest Resources
Yale School of Forestry and Environmental Studies
University of Idaho
U.S. Department of the Interior, Bureau of Land Management, Alaska Fire Service

Abstract

Cronan, J; McKenzie, D; Olson, D. 201X. Fire regimes of the Alaskan boreal forest. Gen. Tech. Rep. PNW-GTR-XXX. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. XXX p.

Fire regimes are described for the boreal forest of Alaska. Information was compiled from published and unpublished work and represents a comprehensive literature review. Contemporary fire regimes are discussed based on six components: ignitions, seasonality, intensity, fire size, severity, and fire frequency. Fire regimes are stratified by major vegetation types present within the boreal forest including black spruce forest, white spruce forest, hardwood forest, and tundra. Multi-scale temporal patterns of fire regimes are presented and future changes to fire regimes and impacts on the boreal forest are discussed.

Keywords: fire regime, fire history, boreal forest, Alaska, literature review.

Contents

Chapter 1: Introduction

History of Fire in Boreal Alaska

The boreal fire regime of interior Alaska was summarized succinctly some 50 years ago when a pioneering ecologist in Alaska said “In one respect all forest trees in the Alaskan interior are similar; they are killed by severe fires” (Lutz 1956). While today that statement remains largely true (French et al. 2008, Johnson 1992a, Murphy et al. 2008), a great deal has been learned about the patterns of fire in the intervening decades. Most importantly, research has revealed the complexity of the disturbance regime and shown important changes in the characteristics of fire over time and space that are influenced by a number of environmental factors.

Once viewed as a relatively simple system linked to irresponsible human actions (Lutz 1956) and as a cause of environmental degradation (Hutchison 1968), fire was rapidly accepted as a disturbance integral to ecological function (Rowe and Scotter 1973, Viereck 1973, Viereck and Schandelmeier 1980), which has continually shaped and been shaped by the natural and human environment since the Laurentide Ice Sheet, and its influence on Alaska’s climate, began receding some 20,000 years ago (DeWilde and Chapin 2006, Hu et al. 2006).

The modern day fire regime in boreal Alaska was established *ca.* 5500 BP (calendar years before Common Era [CE] 1950) (Hu et al. 2006). Since that time, fire has been integral to the ecology of boreal forests. The mainly even-aged stand age structure is almost purely maintained by periodic large stand-replacing fires (Johnson 1992a, Ryan 2002). The fire regime interacts with terrain, soil drainage, forest succession, and flooding in riparian areas to create a mosaic of distinct vegetation types distributed in large patches across the landscape (Fastie et al. 2002, Kasischke et al. 2002, Kane et al. 2007, Mann et al. 1995, Mann and Plug 1999, Viereck et al. 1993). Successional pathways (Epting and Verbyla 2005, Johnstone and Chapin 2006, Johnstone and Kasischke 2005, Kasischke et al. 2000), nutrient cycling (Simard et al. 2007, Viereck and Dyrness 1979), forest productivity (Fenton et al. 2005, van Cleve et al. 1983), permafrost (Harden et al. 2006, Viereck and Dyrness 1979), and wildlife habitat (Collins and Helm 1997, Johnson and Rowe 1975, Maier et al. 2005, Nelson et al. 2008) are all influenced by the boreal fire regime.

Resilient plant species have evolved with the stand-replacing fire regime and rapidly recolonize burned areas following fire. Adaptive strategies revolve around prolific seed production timed in various ways to coincide with fire, and regeneration from rootstock (Black and Bliss 1978, Foote 1983, Viereck and Dyrness 1979, Zasada and Gregory 1969). In general, higher severity favors species that regenerate from seed (Dyrness and Norum 1983, Greene et al. 2004, Johnstone and Kasischke 2005). To a large extent, the composition of the boreal forest in Alaska is dependent upon the current fire regime and vice versa (Brubaker et al. 2009; Higuera et al. 2009; Mann and Plug 1999; Rupp et al. 2000a, 2000b); and both are ultimately regulated by climate (Camill et al. 2009, Duffy et al. 2005, Tinner et al. 2008), which varies at multiple temporal scales (Bartlein et al. 1998, Berger and Loutre 1991, Hess et al. 2001, Hu et al. 2003, Papineau 2001). The relative impacts of climatic shifts depend on their magnitude and duration. Acknowledging these impacts is critical to understanding the nature of the boreal fire regime and has implications for appropriate management of natural resources.

The current statewide wildland fire management plan recognizes the ecological importance of fire and strives to maintain natural fire regimes while achieving primary goals of safely protecting life and property against damage from fires (Alaska Wildland Fire Coordinating Group 1998). This management strategy, incorporating tiered response to fire that balances societal, economic, and ecological priorities, was instituted nearly three decades ago (figure 1). The option to let fires burn in remote areas was slowly embraced by large public land-management agencies (Todd and Jewkes 2006), which were developing policy that recognized the ecological value of fire (Pyne et al. 1996). There is little evidence to suggest that earlier suppression policy had a substantial impact on the fire regime of boreal Alaska (DeWilde and Chapin 2006, Drury and Grissom 2008). This is primarily due to the prohibitive costs of firefighting over such a large and remote area and fire intervals that are generally longer than the period of suppression. Current policy has likely averted major continued ecological impact of suppressing fire by allowing for most fire to burn.

Knowledge of the boreal fire regime is integral to successful land management as the boreal forests in Alaska are fire-dependent ecosystems. Accurate spatially and temporally defined estimates of the fire regime are important for monitoring deviations of the current fire regime from reference fire regime conditions and tailoring management strategies to best suit management goals. Well described fire regimes aid land and fire management agencies in anticipating ecological impacts of interannual and decadal shifts in fire activity, assessing risk to sensitive resources, efficient use of suppression resources, evaluating severity, and forecasting future landscape changes.

As anthropogenic climate change begins to influence the fire regime in the boreal forest of Alaska (Kasischke et al. 2010, Senkowsky 2001) and Canada (Gillett et al. 2004), accurately identifying the drivers of fire regimes and potential feedback loops is increasingly important for several reasons (Hu et al. 2006). Alterations to fire regimes and landscapes caused by climate change will undoubtedly have repercussions for land use patterns across the state and management plans must be adaptive to these changes and have thoughtfully conceived strategies for mitigating negative impacts. A well developed framework to describe drivers of fire regimes is a necessary component of ecological models used to forecast the impacts of climate change.

Statement of Purpose

The purpose of this paper is to provide land and fire management agencies with a comprehensive review of the boreal fire regime in Alaska. New fire history studies in Alaska have been produced nearly every year for the past three decades (table 1), but no comprehensive literature reviews addressing fire regimes in Alaska have been published since the early 1980s (Viereck and Schandelmeier 1980). The large number of studies, occasional lack of accessibility, and difficulty of interpreting results from studies with different methodologies can create an informational burden on land managers.

Methods

We focus on publications that describe fire history of the Alaskan boreal forest. Considerable effort was expended to identify all sources of literature including work that falls outside of scientific literature databases such as government reports and graduate thesis work.

We focus on the boreal forest as opposed to other major cover types because fire is an important component of the natural fire regime (Gabriel and Tande 1983). This review is approached from two angles: a traditional literature review and a compendium of all fire history studies conducted in Alaska.

The main portion of the literature review is divided into three parts, Chapters 2-4 (Chapter 1 is the introduction and Chapter 5 is the conclusion). Chapter 2 reviews each component of the Alaskan boreal fire regime. Since fire frequency is most commonly used to describe how fire regimes change over time and space, this component is the focus of following chapters. Chapter 3 describes the spatial distribution of fire frequency in Alaska vegetation types, and Chapter 4 described the temporal variability of fire frequency across multiple scales. The compendium of fire history studies is Appendix 1. Each summary in Appendix 1 contains a visual presentation of pertinent fire history data followed by a short written summary.

The Regional Environment

Land Cover

The landscape of Alaska has been categorized into ecoregions with unique combinations of environmental characteristics (Gallant et al. 1995, Nowacki et al. 2001). Other efforts have yielded high-resolution vegetation maps (Fleming 1997) and an extensive hierarchy of vegetation classification (Viereck et al. 1992). These efforts clearly show four general vegetation zones: barren land, tundra, boreal forest, and coastal rainforest (figure 2). Although these zones intermix at fine scales, topographic barriers form distinct boundaries (Hare and Ritchie 1972). Gallant et al.'s (1995) ecoregions are best suited to describing broad spatial trends of fire regimes across Alaska and will used throughout this report (figure 2).

Boreal forests cover 32 percent of Alaska's land area (Fleming 1997) and are concentrated in the central and eastern portions of interior Alaska (Gallant et al. 1995, Nowacki et al. 2001). This includes the Interior Forested Uplands and Lowlands, Interior Highlands, Interior Bottomlands, Yukon Flats, and Ogilvie Mountains Ecoregions, which will be collectively referred to as interior Alaska. Other notable areas of boreal forest are found in the Cook Inlet and Copper Plateau Ecoregions. The boreal forests in Alaska are part of a circumpolar biome that has relatively low biodiversity (Hillebrand 2004). There are just six tree species in the boreal forest of Alaska: black spruce (*Picea mariana* (Mill) Britton, Sterns, & Poggenb.), white spruce (*Picea glauca* (Moench) Voss), tamarack (*Larix laricina* (Du Roi) K. Koch), quaking aspen (*Populus tremuloides* (Michx.)), paper birch (*Betula papyrifera* (Marsh.)), and balsam poplar (*Populus balsamifera* (L.)). All species are common in mixed stands and all species except tamarack commonly form monotypic stands. Treeline is approximately 800 m in central and eastern interior Alaska.

Tundra occupies 55 percent of the land area in Alaska (Fleming 1997). The primary domain of tundra in Alaska is to the north and west (Gallant et al. 1995, Nowacki et al. 2001). Alpine tundra is also widespread throughout central and eastern interior Alaska, although boreal forest is the predominant cover type; alpine tundra occupies 27 percent of the landscape of this region (Calef et al. 2005). Vegetation types are divided into two general categories in tundra: those dominated by shrubs and those dominated by herbaceous or graminoid species. Further divisions are based on site moisture, species composition, and shrub stature (Viereck et al. 1992).

Barren land confined to the high peaks of Alaska makes up about 8 percent of the total land area (Fleming 1997). Barren lands are at altitudes with climate that is too harsh for vegetation or are occupied by ice fields. Land cover types are rock, perennial snow, and glacial ice. The largest areas of barren lands occur in the Alaska Range, Wrangell Mountains, and Pacific Coastal Mountains Ecoregions (Gallant et al.

1995). There is no intermixing between boreal forest and barren lands as the tundra vegetation zone separates the two along an altitudinal gradient.

The coastal rainforest is the smallest of the vegetation zones in Alaska and occupies about 4 percent of the landscape (Fleming 1997). Coastal rainforest occur on a narrow strip of low-elevation coastal land that arcs from the eastern end of Kodiak Island eastward throughout southeast Alaska and falls within the Coastal Western Hemlock-Sitka Spruce Forests Ecoregion (Gallant et al. 1995). The zone is dominated by western hemlock (*Tsuga heterophylla* (Raf.) Sarg.), Sitka spruce (*Picea sitchensis* (Bong.) Carrière), and mountain hemlock (*Tsuga mertensiana* (Bong.) Carrière), but a number of other tree species occur, primarily in southeast Alaska. With the exception of a transitional gradient along the eastern coast of the Kenai Peninsula there is little interface between the coastal rainforest and boreal forest in Alaska.

Climate

Being at high latitude, the climate of Alaska is cold, but, as mean annual precipitation and temperature maps in figure 3 illustrate, large differences in climate are apparent across the state (Fleming et al. 2000). These differences are a result of circulation patterns and topography, which has an especially strong effect on the distribution of precipitation (Mock et al. 1998).

During autumn and winter, the counterclockwise circulation around the Aleutian Low funnels moisture towards southeast Alaska (Fleming et al. 2000). The Alaska, Wrangell, and Pacific Coastal Mountains have a strong influence on the spatial extent of these storms; orographic uplift amplifies rates of precipitation in coastal areas while exerting a strong rainshadow effect on the leeward slopes (Mock et al. 1998). The gulf coast of Alaska receives the highest rates of precipitation during the autumn; the maritime influence also produces the warmest winter temperatures in Alaska in this region (Fleming et al. 2000). Within interior portions of the state winter temperatures are much colder, notably at higher elevations and along interior valleys (Fleming et al. 2000). Precipitation is light and monthly precipitation is lowest during winter and spring for areas north of the Alaska Range (Fleming et al. 2000).

During summer the strong pressure gradient between the Gulf of Alaska and interior portions of the state weakens and weak low pressure systems become more common in middle and high latitudes (Serreze et al. 1993). The weakening pressure gradient reduces precipitation along the Gulf coast of Alaska, although rates are still higher than in the rest of the state (Fleming et al. 2000). Precipitation in interior Alaska and the Arctic regions is highest during summer, and the main source of rainfall is convective activity (Serreze et al. 1993, Slaughter and Viereck 1986). The warmest temperatures in the state occur along interior valleys and the coldest temperatures occur along the Arctic coast and at high elevations (Fleming et al. 2000).

Central and eastern portions of interior Alaska have a strongly continental climate (Viereck et al. 1986) and this climate regime is typical of boreal ecosystems (Pojar 1996). The mean annual temperature in Fairbanks is -3.5 ºC in January (the coldest month) and 16 ºC in July (the warmest month); annual precipitation is 286 mm and the wettest months of the year are July and August (Slaughter and Viereck 1986). The highest levels of solar insolation occur in central and eastern interior Alaska (Hare and Ritchie 1972) and absorption is strongly controlled by slope, with soil on south-facing slopes becoming much warmer than north-facing slopes. Differential soil heating, alluvial activity, and elevation gradients are strong drivers of vegetation distribution in the boreal forest (Slaughter and Viereck 1986).

Another impact of the continental climate of interior Alaska is the generation of summer thunderstorms. Air-mass thunderstorms are common and are produced by intense solar heating, atmospheric instability, and local scale weather patterns (Dissing and Verbyla 2003, Reap 1991). Lightning-strike data shows that these storms are strongly focused in central and eastern portions of interior Alaska and peak seasonal activity occurs in June and July (Biswas and Jayaweera 1976, Reap 1991). Local-scale weather patterns are influenced by vegetation and topography (Chambers and Chapin 2002, Dissing and Verbyla 2003).

Fire Regimes

Teleconnection indices, including the Eastern Pacific (EP), Arctic Oscillation (AO), Pacific Decadal Oscillation (PDO) and Southern Oscillation Index (ENSO), exert strong influence on annual and decadal climate (Hess et al. 2001, Papineau 2001), which in turn drives fire activity in boreal Alaska (Duffy et al. 2005, Hess et al. 2001).

Statewide, fire activity is concentrated within the boreal forest of interior Alaska (Gabriel and Tande 1983) (figure 4). Although this area only occupies 30 percent of the state, 96 percent of the fires for the period 1960-99 occurred here (Kasischke et al. 2002). Both the boreal fuel complex and dry and warm summer weather patterns support high fire activity in this region (figures 2 and 3). High loading of available fine fuels in both the surface and canopy layers is joined by extensive ladder fuels to encourage high-intensity crown fires (Hely et al. 2000, Norum 1982, Schimmel and Granstrom 1997, Van Wagner 1983, Viereck 1983). These fuel conditions are contiguous across the landscape with few topographic barriers, thus setting the potential for large fires (Burton et al. 2008, Parisien et al. 2006). During early summer extended periods of clear dry weather and poor night time recovery of relative humidity cause widespread drying of fine surface fuels. Lightning frequency also reaches its peak during this time (Dissing and Verbyla 2003, Reap 1991). During years with extended periods of dry and warm weather during the summer, large areas are likely to be burned (Duffy et al. 2005; Hess et al. 2001; Larsen 1996; Macias Fauria and Johnson 2006, 2008; Xiao and Zhuang 2007).

Outside the boreal forest, the synergy between fuels and climate to produce frequent, large fires does not occur to the same extent in the tundra in interior Alaska and the fire cycle is substantially longer (Kasischke et al. 2002). Fires comparable in size to boreal fires have been documented in tundra (Jones et al. 2009), but these fires occur infrequently and burn as low-intensity surface fires (Wein 1976). Tundra is generally considered to be less flammable than boreal forest, and climate is cooler and cloudier (Fleming et al. 2000, Trigg 1971). Differences in tundra fire regimes in Alaska do exist and these differences are likely caused by climate. The main dichotomy is between tundra that is part of interior Alaska (i.e. the Seward Peninsula and Noatak River Valley), which has a fire cycle between 250-700 years (Gabriel and Tande 1983, Kasischke et al. 2002, Racine et al. 1985), and tundra dominated ecoregions to the north and south, which have a fire cycle well over 1000 years (Gabriel and Tande 1983). Climate may explain this difference as the western coast of Alaska has a transitional climate between warm and dry continental climate and maritime climate, whereas the climate during the growing season is substantially colder north of the Brooks Range and wetter in southwest Alaska (Trigg 1971).

Fires do not occur in the two other vegetation zones, barren lands and coastal rainforest, due to wet weather year round, and in the case of the former, a lack of fuels. Fuels are abundant in the coastal rainforest and occasionally small fires will occur during dry periods, but these events are a negligible component of the disturbance regime (Barney 1971).

Chapter 2: Alaskan Boreal Fire Regimes

Reconstructing and Interpreting Fire Regimes

Reconstructing fire regimes is crucial to managing fire-dependent ecosystems (Bergeron et al. 2002, 2004a; Chapin et al. 2008; Johnson et al. 1998; Ward and Mawdsley 2000). Detailed descriptions of fire regimes provides guidance for fire management programs and are important to a number of other areas of resource management including wildlife, fisheries, hydrology, water quality, air quality, silviculture, and carbon cycling (DeBano et al. 1998; Wein and MacLean 1983).

The concept of a fire regime presents a framework to quantify and standardize fire characteristics across multiple temporal and spatial scales. However, the definition has shifted throughout time (Krebs et al. 2010) and this report will adhere to the contemporary definition (Bond and Keeley 2005) commonly applied to the North American boreal forest (Johnson 1992a, Ryan 2002). There are six components of the fire regime: ignitions, seasonality, intensity, fire size, severity, and frequency.

Though all components of boreal fire regimes are studied, current knowledge is weighted towards fire frequency (Morgan et al. 2001). The reason for this is simply that while evidence of historic fires is available, it provides little evidence except for the approximate date of the fire, thus only fire frequency can be estimated.

The dependence on fire frequency as a default measure has been somewhat fortunate because fire frequency in the boreal forest is important from two standpoints. First, ecologically, fire frequency is an important driver of species composition (Chen et al. 2009, Mann and Plug 1999, Ryan 2002, Sirois and Payette 1991, Weir et al. 2000), patterns of succession (Brassard and Chen 2006, DeBano et al. 1998, Frelich and Reich 1995, Johnstone and Chapin 2006), regeneration strategies (Gauthier et al. 1996), stand productivity (Simard et al. 2007), and landscape vegetation dynamics (Calef et al. 2005, Rupp et al. 2000b). Second, from the perspective of communicating concepts, fire frequency has been used successfully to compare boreal fire regimes across spatial and temporal scales (Cyr et al. 2007; Jiang et al. 2009; Kasischke et al. 2002, 2010; Larsen 1997; Weir et al. 2000), develop frameworks that describe interactions between fire, biotic, and abiotic factors (Hu et al. 2006), and to quantify the affects of climate change on fire regimes (Bergeron 1998, Calef et al. 2005, Wotton et al. 2010). As such, this report will review all components of the boreal fire regime, but the focus will be on fire history studies (i.e. assessments of fire frequency) given their continuing emphasis.

Three types of data are used to quantify fire frequency in the boreal forests: fire perimeter maps, tree ring/fire scar data (dendrochronological), and lake-bottom sediment cores (paleoecological) data (Kasischke et al. 2006, Wein and MacLean 1983). Each has unique applications and limitations and a trade-off exists between the temporal and spatial breadth of the data (Johnson 1992b, Murphy et al. 2000, Pyne et al. 1996, Swetnam et al. 1999). All three types of data have been widely used to evaluate fire frequency, though over the past decade paleoecological records and fire perimeter databases have been favored over dendrochronological records in fire history research (table 1).

Like fire frequency, severity is also an ecologically important component of the boreal fire regime, especially its impact on long-term succession (Johnstone et al. 2010a, Kasischke et al. 2000, Johnstone and Kasischke 2005, Payette 1992), and interactions between climate change, severity, and vegetation patterns (Barrett et al. 2010, Bernhardt et al. 2010). Unlike fire frequency, it is difficult to reconstruct historic trends of fire severity and only a few studies have attempted to do so for the boreal forest (e.g.

Camill et al. 2009). Early severity research focused on the interactions between severity and ecosystem dynamics (Dyrness and Norum 1983, Foster 1985, Johnston and Elliott 1998, Neary et al. 1999, Schimmel and Granstrom 1996, Sirois and Payette 1991, Viereck 1983, Viereck and Dyrness 1979), while research over the past decade has focused on high resolution quantitative assessment of severity and at large spatial scales via remote sensing (Allen and Sorbel 2008, Barrett et al. 2010, Boby et al. 2010, Duffy et al. 2007, Epting and Verbyla 2005, Epting et al. 2005, French et al. 2008, Hall et al. 2008, Hoy et al. 2008, Kasischke et al. 2008, Murphy et al. 2008, Soverel et al. 2010, Verbyla and Lord 2008, Wulder et al. 2009).

The remaining components have important applications for research and management of boreal fire regimes. However, their representation in the literature is relatively weak, and medium to long term historical reconstructions are limited or non-existent. Of these components, fire size data is perhaps most reported and is generally included in spatially explicit fire history studies (e.g. Barney 1971; Gabriel and Tande 1983; Kasischke et al. 2002, 2010). Seasonality data is occasionally constructed from fire scar data but resolution is poor (e.g. De Volder 1999), and reconstructions are generally limited to the contemporary fire regime (e.g. DeWilde and Chapin 2006, Kasischke et al. 2002). Data for ignitions and intensity are limited to the contemporary fire regime. Several studies have examined the spatial and seasonal patterns of lightning strikes (Biswas and Jayaweera 1976, Dissing and Verbyla 2003, Reap 1991), and the distribution of lightning- and human-caused ignitions (DeWilde and Chapin 2006). Collecting intensity data is time consuming and resource intensive and studies are limited to a relatively small number of data points in general fuel types (Ryan 2002).

Ignitions

In the boreal forest of Alaska, as with the rest of the boreal forest in North America, the vast majority of acreage is burned by lightning-caused fires. Estimates in Alaska range from 78-97 percent based on analysis of fire-management records (Barney 1971, DeWilde and Chapin 2006, Gabriel and Tande 1983), a range that is similar to values calculated for the Canadian boreal forest (Johnson and Rowe 1975, Rowe et al. 1975, Stocks et al. 2002) and the North American boreal forest as a whole (Kasischke and Turetsky 2006). It is clear that lightning-caused fires are the source of most area burned, but conclusions regarding the primary source of ignition for number of fires are ambiguous. Fire-management records from the Canadian boreal forest show that most fires are started by lightning (85 percent) (Johnson and Rowe 1975, Rowe et al. 1975), whereas in Alaska the reverse is true. Human-caused fires (62-70 percent) are more frequent than lightning-caused fires, although these data may be biased toward human-caused fires due to early beliefs that most or all fires were started by people (Barney 1971, Gabriel and Tande 1983).

There is a strong regional-scale association between lightning strike density (Reap 1991) and lightning-caused fire distribution (Gabriel and Tande 1983) (figures 3c and 5a, respectively). Paleoecological data suggests that relative to neighboring boreal forests on the Kenai Peninsula, lightning-caused fires have been more common in interior Alaska for at least the past 1000 years (Lynch et al. 2002). Within interior Alaska, lightning strike density increases from west to east (Reap 1991) which may contribute to a west to east increase in fire frequency (Kasischke et al. 2002).

Local-scale features such as terrain and vegetation also influence spatial variation of lightning density. For instance, lightning strikes are concentrated at higher elevations within interior Alaska (Reap 1991) and in spruce dominated stands (Dissing and Verbyla 2003). However, at the local scale there is not necessarily an association between lightning strike density and fire frequency. This is shown by the lower incidences of fire above treeline despite high lightning strike density. This indicates a dynamic relationship with other factors (e.g. fuel properties and precipitation), that determines fire frequency (Kasischke et al. 2002). Below treeline, data do indicate a possible increase in fire activity with increasing elevation (Fastie et al. 2002) but it is not clear if this is due to higher lightning frequency or changes in the flammability of the fuel complex. Among boreal vegetation types the higher density of lightning strikes within boreal spruce is associated with increased ignitions (DeWilde and Chapin 2006).

Differential albedo among vegetation types may explain the higher probability for lightning-caused fires in boreal spruce. Boreal spruce has a relatively low albedo (Baldocchi et al. 2000, Chapin et al. 2000) and differences in energy exchange between boreal spruce and other vegetation types can in turn promote more convective activity and thunderstorms over areas where boreal spruce is present (Chambers and Chapin 2002, Dissing and Verbyla 2003).

In interior Alaska, most lightning-caused fires start within the typical range of the fire season: June through mid July (DeWilde and Chapin 2006), which matches the seasonal distribution of lightning strikes (Reap 1991). Thunderstorms in Alaska are most common following the summer solstice because solar radiation reaches peak levels, which in turn generates substantial convective activity through land mass heating. This, combined with an unstable atmosphere and low wind shear, creates optimal conditions for thunderstorms in the absence of any major weather systems (Biswas and Jayaweera 1976). These are referred to as air mass thunderstorms and are generated entirely by local scale weather processes. Synaptic scale weather patterns including the passage of frontal boundaries or low-pressure systems can also generate thunderstorms (Biswas and Jayaweera 1976).

Human-caused fire ignitions have different spatial and temporal distributions from those caused by lightning (see figures 5 and 6). Throughout Alaska, their contemporary distribution is, perhaps unsurprisingly, associated with major roads and population centers (Gabriel and Tande 1983). There is a long history of human-caused fires in Alaska. Some Athabaskan groups historically used fire to combat insects, as an aid in hunting, signaling, and preparing timber for uses such as fence building and firewood (Lutz 1956, Natcher et al. 2007). Human-caused fires during pre-settlement times were most likely concentrated in western interior Alaska where Athabaskan groups used fire as a management tool (Natcher et al. 2007). Studies throughout the boreal forest show that the frequency of human-caused fires increased during periods of European settlement in northern Europe (Niklasson and Granstrom 2000), Canada (Weir et al. 2000), and Alaska (De Volder 1999, Fastie et al. 2002, Lynch et al. 2002). During the settlement periods, when roads in Alaska were non-existent, human-caused fires were likely concentrated along traditional travel routes such as rivers and population centers (Gabriel and Tande 1983).

Human-caused fires are concentrated towards the beginning of the fire season during May and early June, and occur in fuel types with fine, sun-exposed dead fuels that are most flammable after snowmelt, including leaf litter in deciduous forest and dead grasses in meadows (DeWilde and Chapin 2006).

Seasonality

In a book on the role of fire in northern ecosystems, the authors assert that “no one has been able to define a satisfactory ‘normal’ fire season” (Barney and Stocks 1983). Their quote highlights the large year-to-year variability of fire seasons in the boreal forest (figure 7). This variability also holds true for Alaska. For example, in 2004 and 2005 the fire season extended from May through early September and well over 1.6 million ha burned each year. In contrast, in 2006, less than 121 000 ha burned and the fire season was short lived, restricted mostly to June (Alaska Interagency Coordination Center 2005, 2006). Notwithstanding this variability, the ‘average’ fire season in Alaska occurs from late May through early June. During this period, the weather tends to be dry and warm with frequent dry lightning. Combined with long periods of solar radiation that prevent nighttime recovery of relative humidity, fine fuel moisture remains low for extended periods of time and will often easily support the start and spread of wildfire. For the period 1995-2001, 76 percent of fires in interior Alaska occurred from June 1 through July 15 (Kasischke et al. 2002). However, during years when the area burned exceeded the average by 50 percent or more, the fire season extended into July and August, as was the case in 2004 and 2005. Dendrochronological data from the Kenai Peninsula for the period 1708-1947 show that about half the fires occur during the early growing season (defined as from May through mid June) while the other half occur in the mid to late growing season (defined as from July through August) (De Volder 1999). This indicates a different fire season between the boreal forests of interior Alaska and the Kenai Peninsula.

Late-running fire seasons are often caused by dry and warm weather that extends into July and August (Larsen 1996). Typically the fire season ends in mid July when low pressure systems begin to replace blocking high pressure systems and an increase in relative humidity and rainfall increase fuel moisture and restricts fire growth for the remainder of the growing season. Late-running fire seasons are associated with moderate to strong positive phases of El Niño, as is an increase in lightning activity (Hess et al. 2001) and seasonal area burned is both above average and concentrated in mid to late summer.

Severity, primarily determined by surface organic layer consumption, in the Alaskan boreal forest (Kasischke et al. 2008), increases over the course of the fire season (Kasischke and Johnstone 2005). Several factors influence this interaction including seasonal weather patterns, active layer depth, surface organic layer depth, and soil drainage (Bourgeau-Chavez et al. 2007, Kasischke and Johnstone 2005).

Vegetation and terrain can often create different fire seasons within the same geographic area (Beaty and Taylor 2001, Heyerdahl et al. 2007, Taylor 2000). This has not been examined in the boreal forests of Alaska but is unlikely due to gentle terrain and largely continuous forests dominated by a relatively small number of species.

A possible exception is hardwood stands, especially on south facing slopes. Research on similar forest types in the boreal forest of Canada has shown that hardwood stands tend to be more flammable in the early season compared with conifers (Hely et al. 2001). Hardwood stands are known to be most flammable before leaf out and after leaf drop due to increased solar radiation on the forest floor and drying of the litter layer (Van Wagner 1983). While this is most likely on steeper south facing slopes where early season solar radiation is greatest and snowmelt occurs earliest, variation of fire season throughout the region due to this effect is likely minimal, as steep south-facing slopes with hardwood forests do not make up a significant percentage of the land area.

Fire behavior is largely uniform over the course of a typical fire season with most acreage burned by passive and active crown fires. Possible variability includes the early season, when fires carried through hardwoods burn as low-intensity surface fires, and during large fire years when a greater proportion of higher-intensity active crown fires are likely due to drier fuels, especially in the duff layer.

Climate change is an additional influence on seasonality throughout the boreal forests of North America. Documented changes in the boreal climate of North America include a warming of the average temperature by 1-1.6 ºC since the early 1970s (Hansen et al. 1996), rising average decadal area burned (Gillett et al. 2004, McCoy and Burn 2005), higher frequency of large fire years, a longer fire season, and more late-season fires (Kasischke and Turetsky 2006). Continued warming will likely further lengthen the fire season and increase annual area burned across the North American boreal forest (Flannigan et al. 2001, 2005; Wotton and Flannigan 1993).

Intensity

Intensity refers to the magnitude of fire behavior and can be quantified by fireline intensity, defined as the rate of release of energy along a linear fire front and is reported as kilowatts per linear meter (kW/m). Fireline intensity is a useful gauge of fire behavior because it incorporates other common measures of fire behavior including rate of spread (*ROS*) and flame length. Calculating fireline intensity (*FLI*) requires two variables: heat per unit area (*HPA*) and *ROS* (Byram 1959). The equation is given by:

[2.1] $$FLI = \frac{HPA * ROS}{60}$$

Where HPA equals the heat content of fuel (kJ/kg) times the mass of fuel consumed (kg/m^2). Unfortunately measuring these variables is difficult and resource intensive so there are relatively few quantitative studies that directly measure fireline intensity.

There have been no field-based estimates of fireline intensity in the boreal forest of Alaska, but multiple experimental burn projects conducted throughout Canada used this technique to measure the input parameters for Byram's (1959) equation and estimate fireline intensity for a range of boreal fuel types and weather conditions. These fuel types and associated fire regimes are broadly similar to those in the Alaskan boreal forest. The results of these studies are frequently used to represent the range of fire behavior for the North American boreal forest which shares a high-intensity surface fire to crown fire regime (Bourgeau-Chavez et al. 2000). Fire behavior modeling for Alaska fuel types shows that estimated fireline intensity data for Canadian forests generally applies to the Alaskan boreal forests (Cronan and Jandt 2008).

Fireline intensity in the North American boreal forests is quite high, a consequence of frequent crown fires. That said, fireline intensity still can vary widely in boreal forests and both ground and surface fires are common occurrences. Fireline intensity ranges from less than 10 kW/m for smoldering, to greater than 100 000 kW/m for wind driven crown fires (figure 8).

In each of the studies referenced below, fuel consumption and rate of spread were inputs for the standard equation (Equation 2.1 -- above) to estimate fireline intensity (Byram 1959).

Fireline intensity for ground fires is near the limit of smoldering consumption, about 10 kW/m (Van Wagner 1983). Estimated fireline intensity for surface fires was generally greater than 500 kW/m (Bourgeau-Chavez et al. 2000; Byram 1959; Quintillo et al. 1977; Stocks 1987, 1989). Passive crown fires produced fireline intensities of 500-3000 kW/m (Alexander et al. 1991, Quintillo et al. 1977, Stocks 1989) while active crown fires generated fireline intensities over a large range of 4 000-100 000 kW/m (Alexander et al. 1991, Quintillo et al. 1977, Stocks 1989, Stocks et al. 2004). To provide a sense of scale, fireline intensities above 350-500 kW/m will likely overrun hand constructed fire breaks and fireline intensities above 2000-4000 kW/m may be resistant to any type of suppressive actions (Alexander and Lanoville 1989, Rothermel 1983).

As mentioned above, the fireline intensity estimates from experimental burns in Canada were conducted in fuel types with ground fuel loadings that are low relative to analogous fuel types in Alaska (i.e. black spruce-feathermoss and white spruce-feathermoss). During periods of extended drought the ground fuels can dry, and depending on whether or not the ground fuels consume in the flaming front or during residual consumption that may increase fireline intensity substantially. Modeling by Cronan and Jandt (2008) does not account for potential contributions by ground fuels.

Another significant fuel type in the Alaskan boreal forest is the boreal hardwood forest. This fuel type is generally most flammable before leaf-out in spring and following leaf-drop in the autumn (Van Wagner 1983). Estimated fireline intensity for trembling aspen in central Alberta before leaf-out in the spring was 15-581 kW/m, although another experimental fire conducted several years later at the same location, but under high-severity weather, had an estimated fireline intensity of 4200 kW/m (Quintillo et al. 1991). These intensities are still much lower than peak estimated fireline intensities in coniferous boreal fuel types (figure 8), though from a fire suppression standpoint; this study shows the potential for deciduous boreal forest types to produce overwhelming fire behavior under severe weather scenarios. Fireline intensity potential in boreal deciduous fuel types is generally low because the understory and overstory fuel strata are not flammable and also provide dense shade, maintaining high moisture content of fine dead surface fuels during most the fire season. During leaf free periods in the early season, available fine fuels are generally restricted to the leaf litter, which limits fire behavior to light surface fuels. Direct evidence of lower flammability in deciduous boreal fuel types has been supported by evidence anecdotal from experimental fires in Alaska (Hinzman et al. 2003) and Canada (Alexander and Lanoville 2004).

Fire Size

Fire size simply refers to the area within a fire perimeter and the distribution of these areas for a fire regime (Ryan 2002). Multiple environmental factors control fire size and understanding their effects is an important key to describing ecological patterns at multiple spatial scales in fire-dependent ecosystems (DeLong and Kessler 2000, DeLong and Tanner 1996, Hansen et al. 1991, Turner et al. 1997, Wang and Cumming 2010). Fire management agencies in Alaska have recorded fire perimeters since the inception of organized fire suppression in 1939, and all records are maintained in a single statewide database (Murphy et al. 2000). The quality of records has improved over time with the introduction of aircraft, GPS/GIS technology, and satellite imagery (Kasischke et al. 2002, Murphy et al. 2008).

Mapping the spatial extent of pre-suppression era fires involves considerably more effort and reliance on more subjectively interpreted data including dendrochronological records and aerial photos to delineate fire perimeters. These techniques have been used at relatively small scales in Alaska (De Volder 1999, Fastie et al. 2002) and similar, if not more extensive, studies have been conducted elsewhere in the boreal forest (Bergeron et al. 2001, Foster 1983, Masters 1990, Niklasson and Granstrom 2000, Weir et al. 2000).

Fire perimeter data has been utilized to study spatial patterns during the past 40 years in Alaska (Barney 1971; DeWilde and Chapin 2006; Gabriel and Tande 1983; Kasischke et al. 2002, 2010), and more recently, have been used in larger meta-analyses of boreal fire patterns (Kasischke and Turetsky 2006, Soja et al. 2007, Xiao and Zhuang 2007).

The average area of fires in the Alaskan taiga is 870 ha (Gabriel and Tande 1983), though this belies the large variability of fire sizes. Fire size distributions at the regional and continental scale for the North American boreal forest, including Alaska reflect an exponential distribution (Cumming 2001a), where most fires are small (less than 400 ha) but the majority of land is burned by large fires (greater than 400 ha) (Bergeron et al. 2002, DeWilde and Chapin 2006, Kasischke et al. 2002, Stocks et al. 2002).

The largest fires occur primarily during large fire years and burn a disproportionate amount of the landscape (Cumming 2001a, Foster 1983, Stocks et al. 2002). For example, in Alaska large fire years occurred 17 percent of the time in and accounted for 68 percent of the area burned (Kasischke et al. 2010). During average fire years the distribution of area burned among size categories shifts, and smaller fire size categories burn a larger proportion of annual area burned (Kasischke and Turetsky 2006, Kasischke et al. 2002).

The frequency of large fires and fire years has not been steady over time (figure 7). For 50 years of data starting in 1940, frequency of very large fires in Alaska (greater than 50 000 ha) was greatest during the 1940's, 1950's, 1990's and 2000's (Kasischke et al. 2010). Large fire years in the North American boreal forest are influenced primarily by teleconnection indices that drive annual and decadal scale climate variability (Duffy et al. 2005, Hess et al. 2001, Skinner et al. 2002) and large fires are a primary control of fire frequency (Johnson et al. 1990, Payette et al. 1989). The 2004 and 2005 fire seasons are an example of this interaction. Together they accounted for the majority of the area burned in the most recent decade and they decreased the contemporary fire cycle of interior Alaska from 216 years (1950-99) to 156 years (1950-2009) (Kasischke et al. 2010).

Though climate is the main driver of fire activity in boreal forests of North America (Bergeron 1991, Bessie and Johnson 1995, Johnson et al. 1990), terrain and vegetation factors are also a determinant of spatial burning patterns (Cumming 2001b, Dansereau and Bergeron 1993). In boreal forests of Canada and northern Europe, higher waterbreak (i.e. wetlands, ponds, lakes) density resulted in smaller more numerous fires (Dansereau and Bergeron 1993, Hellberg et al. 2004) but these features did not have a statistically significant impact on spatial patterns of fire in the boreal forests on the Kenai Peninsula of Alaska (De Volder 1999). The influence of vegetation on fire size appears to be linked with flammability in interior Alaska where large fires were most frequent in vegetation types with the highest fire frequency (DeWilde and Chapin 2006). These spatial relationships have also been supported by spatially explicit modeling results for interior Alaska (Rupp et al. 2000a).

There are also multiple interactions with between fire size and severity. Large fires have more heterogeneous distribution of severity (Burton et al. 2008, Duffy 2006), higher density of unburned islands (Eberhart and Woodard 1987), and higher average severity (Duffy et al. 2007). Data from interior Alaska contradicts unburned island results presented by Eberhart & Woodard (1987) suggesting additional factors may influence the relationship between fire size and density of unburned islands (Kasischke et al. 2010).

A common trend across the boreal forest of North America absent in Alaska (and Yukon Territory) is the latitudinal decrease in average fire size (Payette et al. 1989, Rowe et al. 1974). This occurs throughout most of Canada because flat terrain allows for a gradual climate-driven transition from high to low fire activity that accompanies the transition from boreal forest to tundra. In Alaska, the Brooks Range creates a topographic barrier between the two biomes and the transition is abrupt with no latitudinal trend in fire size (Bourgeau-Chavez et al. 2000).

Human activities also have an impact on fire size distribution in the boreal forest. In Canada (Lefort et al. 2003) and Alaska (DeWilde and Chapin 2006, Gabriel and Tande 1983) populated areas have a larger number of fires, but smaller average size. From 1992-2001 human-caused fires only accounted for 4.6 percent of the area burned in interior Alaska and of these fires 78 percent were less than 0.4 ha (DeWilde and Chapin 2006). Despite the larger number of fires, the overall effect of populated areas on the fire regime is decreased fire frequency (Cumming 2005, DeWilde and Chapin 2006, Martell and Sun 2008). Average fire size in populated areas is smaller for several reasons. In Alaska fire policy dictates aggressive suppression in populated areas and concentrates suppression resources in these areas. Additionally, better road access decreases response time for fire suppression personnel (Todd and Jewkes 2006). Human-caused fires also disproportionately occur in early spring when fire danger is low fuel, and in low flammability fuel types; thus, they are more easily suppressed (DeWilde and Chapin 2006).

In summary, fire size is primarily driven by climate with large fires occurring during exceptionally dry and warm years. However, spatial variables such as topography, vegetation, and land use all have more subtle influences on fire size. Furthermore, changes in these factors over time can have lasting impacts on landscape-scale ecological patterns (Weir et al. 2000).

Severity

Severity is the environmental impact of wildland fire and is sometimes grouped with intensity under the broader heading of fire magnitude (Agee 1993, Ryan 2002). Generally, this is practical because intensity directly impacts severity. However, in the boreal forest of Alaska the two measures are less related (Kasischke et al. 2006, Wulder et al. 2009) and they have been separated in this report. There is no commonly used definition of severity (Key and Benson 2006); a consequence of the myriad impacts fire has on the environment (Neary et al. 1999). Thus, the definition changes among studies and measures are often indices of quantitative and qualitative variables tailored to evaluate specific impacts (French et al. 2008, Lentile et al. 2006, Ryan 2002). For the sake of convenience and comparative purposes, severity values are frequently partitioned into categories (Lentile et al. 2006) and in here we will refer to severity in two general categories, low and high.

Measures of overstory damage and mortality are often important indicators of severity in forested ecosystems. In Alaska overstory mortality is nearly 100 percent and these measures have little bearing on the impact of fire (Kasischke et al. 2008). Thus they are often excluded from proposed severity indices which instead focus on consumption within each fuel strata (Ryan 2002, Viereck and Schandelmeier 1980).

Despite the lack of a common definition for severity, some standardized measures have been introduced. Perhaps the most commonly used is the composite burn index (CBI), a standardized field based index (Key and Benson 2006). CBI has been used extensively in Alaska, although evaluations of effectiveness are contradictory (Boby et al. 2010, Kasischke et al. 2008). Remotely sensed evaluations of severity are often used to aid post-fire restoration response. Widely used standardized indices include the normalized burn ratio (NBR), differenced NBR (dNBR), and relative differenced NBR (RdNBR). Other indices exist, but they are less effective than NBR in boreal Alaska (Epting et al. 2005). NBR is based on a single post-fire image while dNBR is the absolute difference between pre- and post-fire NBR images. This technique standardizes spatial variability of pre-fire vegetation types (Key and Benson 2006). RdNBR is similar to dNBR but severity is scaled based on the relative, rather than absolute, differences in pre- and post-fire values (Miller and Thode 2007). To account for concerns with NBR and its variants, customized measures have been recently introduced to focus on the surface organic layer (Barrett et al. 2010, Verbyla and Lord 2008).

Assessments of NBR in interior Alaska have shown good agreement in forested non-wetland boreal vegetation types (Epting et al. 2005, Verbyla and Lord 2008), while assessments of dNBR have, with one exception (Allen and Sorbel 2008), generally been poor (Hoy et al. 2008, Murphy et al. 2008). Assessments in adjacent areas of Canada have shown that NBR and its variants can be well correlated with field measures of severity (Hall et al. 2008, Soverel et al. 2010, Wulder et al. 2009), though these correlations often have substantial caveats limiting the scope of their applicability (French et al. 2008).

A general concept in fire ecology is the inverse relationship between severity and fire frequency (Agee 1993). To an extent, this has been detected in the boreal forest of Alaska. Remotely sensed measures of severity indicate that deciduous forest, which have a high fire frequency relative to boreal spruce (Larsen 1997, Mann and Plug 1999, Yarie 1981), have lower relative measures of severity (Allen and Sorbel 2008, Duffy et al. 2007, Epting and Verbyla 2005, Verbyla and Lord 2008). Studies in Canada show the same relationship across space (Bergeron 1991) and time (Camill et al. 2009) for the boreal forest. However, given the high spatial variability of fire frequency (Kasischke et al. 2002) and the relatively weak relationship between fire frequency and forest type (Drury and Grissom 2008), it is unclear if fire frequency directly impacts severity. As evidence, research building on the work of Bergeron (1991) found that while the relationship between severity and forest type held (Wang 2002), lower severity fires were not statistically correlated with high fire frequency (Bergeron et al. 2004a). These findings are consistent with recent data from Alaska showing that spatial patterns are the main driver of burn severity (Barrett et al. 2010).

The relationship between severity and fire frequency has not been found between other vegetation types in interior Alaska. Despite evidence of lower fire frequency in white spruce relative to black spruce (Quirk and Sykes 1971, Mann and Plug 1999), remotely sensed measures of severity were not significantly different (Allen and Sorbel 2008). Another contrary example is alpine tundra which has a lower fire frequency relative to boreal forest (Kasischke et al. 2002), yet also has lower measures of severity (Epting and Verbyla 2005, Allen and Sorbel 2008).

Other measures of vegetation also have an influence on spatial variability of severity in the Alaskan boreal forest. A positive correlation has been observed between severity and canopy cover; perhaps a consequence of increasing fuel load along a gradient from spruce woodlands to closed canopy forest (Epting and Verbyla 2005). Severity also appears to be more uniform as the density of willow and trembling aspen increases (Verbyla and Lord 2008). This is likely a result of the wide variability of climate driven fire behavior in spruce types as opposed to the muted and narrow range of fire behavior in deciduous shrub and forest types (Alexander 2010, Quintillo et al. 1991).

Following fire in the North American boreal forest, severity may appear high given that near complete overstory mortality is common. This is in fact, not true. As mentioned above, in boreal forest, severity is primarily a function of consumption of fuel strata, primarily of the surface organic layer consumption. Following boreal crown fires the spatial variability of this measure within fire perimeters is high (Bergeron et al. 2002, Dyrness and Norum 1983, Kasischke and Johnstone 2005, Miyanishi and Johnson 2002). Heterogeneity of severity increases with fire size (Burton et al. 2008, Duffy 2006, Eberhart and Woodard 1987), though other studies show otherwise (Kasischke et al. 2010).

Topography also controls severity and studies suggest that as topography becomes more complex the heterogeneity of severity increases and the influence of vegetation wanes (Duffy et al. 2007, Kafka et al. 2001). Along topographical gradients severity is higher on slopes with south facing aspects (Barrett et al. 2010, Kane et al. 2007, Paragi and Haggstrom 2007) and in upland relative to lowland black spruce forest

(Barrett et al. 2010). However, based on the percentage of surface organic matter consumed, severity appears to be uniform between sites with and without permafrost when other variables (i.e. topography and vegetation type) are standardized (Harden et al. 2006).

The spatial patterns of severity illustrate the complexity of landscape processes driving both fire behavior and environmental response to fire (Wulder et al. 2009). For instance, on flat terrain deciduous forests are widely associated with lower severity because the fuel complex supports lower fire behavior, yet in hilly or mountainous terrain, severity is highest on slopes with south aspects where deciduous forest types are most prevalent, most likely these areas are most sensitive to drought and high fuel availability ensures high rates of consumption. Thus, the impact of aspect appears to override the influence of fuels.

With the exception of seasonal trends, temporal trends of burn severity are less well understood. Seasonal trends have been studied more extensively given the applications for fire management strategies. Based on severity data from three fires Kasischke and Johnstone (2005) proposed a seasonal model that accounted for interactions among permafrost, soil texture, depth of the organic layer, and seasonal moisture values. Their model states that there is an uneven response to seasonal drying based on multiple soil properties. More specifically, as the burn season progresses locations with moderate surface organic depths, permafrost, and better soil drainage become more susceptible to higher severity fires (Kasischke and Johnstone 2005).

While the drought code (DC) of the Canadian Fire Danger Rating System would seem like a natural indicator of potential severity, and has been shown to work in boreal forest of Canada (de Groot et al. 2009), it does not perform well in black spruce forests of Alaska due to moisture inputs from thawing permafrost (Kasischke and Johnstone 2005).

Multivariate analysis of severity dynamics generally support single factor relationships identified in studies cited above but show a complex dynamic that operates on multiple temporal and spatial levels to affect severity (Barrett et al. 2010).

Severity dynamics have important implications for ecosystem management and the development of models to forecast the impact of climate change on ecological and biogeochemical processes. These applications require information on the longer term environmental effects of severity, or fire effects. In general, fire effects that have a positive response to severity include site productivity (Dyrness and Norum 1983, Lecomte et al. 2006b, Simard et al. 2007, Zasada et al. 1987), active layer depth, and soil temperature (Kasischke et al. 2000). High severity fires may also alter soil microbe composition (Smithwick et al. 2005) and reduce surviving belowground plant material thereby favoring regeneration from seed sources (Dyrness and Norum 1983, Schimmel and Granstrom 1996, Johnstone and Kasischke 2005).

In the North American boreal forest post-fire recovery is generally self replacement (Barrett et al. 2010, Epting and Verbyla 2005, Viereck 1983). Fire-induced vegetation change occurs less commonly than self replacement and is associated with high severity fires (Johnstone and Kasischke 2005, Johnstone et al. 2010a, Lieffers et al. 1993, Viereck and Dyrness 1979). Figure 9 illustrates departures in forest structure and composition associated with high and low severity fires. Overall heterogeneous severity is important to regional ecological health by creating stand-level habitat diversity and increasing the range of vegetation communities (Burton et al. 2008); another component of severity, unburned islands, serve as important nodes of regeneration in burned landscapes (DeLong and Kessler 2000). However, climate change will likely increase severity (McCoy and Burn 2005, Xiao and Zhuang 2007) which will have major implications for future forest composition and structure (Chen et al. 2009, Johnstone et al. 2010a, Kasischke et al. 2010, Turetsky et al. 2010, Wirth et al. 2008). Forecasted landscape-level vegetation shifts are supported by reconstruction of historic climate induced shifts in the fire regime that precipitated large-scale shifts in vegetation distribution (Tinner et al. 2008, Camill et al. 2009).

Fire Frequency

Fire frequency is the rate fires return to a given area; this is typically expressed as fire cycle or fire return interval (*FRI*) (Ryan 2002). Briefly, fire cycle is the amount of time in years it takes for an area equal to the area being studied to burn and *FRI* is the amount of time in years between two successive fires within a given area. *FRI* is generally reported for a point location and intervals are often averaged and reported as mean *FRI* (*mFRI*). The *mFRI* provides greater spatial and/or temporal depth based on whether the intervals represent multiple points or a single point, or some combination of the two.

Evaluating fire frequency in the North American boreal forest poses several challenges. First, fires have high overstory mortality and fire scarred trees, especially those that record multiple fires, are rare enough that it is not possible to determine fire history for periods greater than two to three fires (Dyrness et al. 1986). Second, climate is a major control of interannual (Johnson et al. 1998) and decadal variability in area burned (Duffy et al. 2005, Hess et al. 2001, Macias Fauria and Johnson 2008, Skinner et al. 2002). This variability, combined with long *FRI*s, mutes any spatial variation and decadal temporal trends (e.g. response of fire regimes to fire suppression policy).

Three types of data are used to quantify fire frequency in the boreal forests of Alaska: fire perimeter databases, dendrochronological records, and paleoecology data. Each type of data has unique applications and limitations. Generally, a trade-off exists between the temporal and spatial breadth of the datasets. Fire perimeter datasets operate at large spatial and short (decadal) time scales while paleoecological data operates at small spatial and long (millennial) time scales, and dendrochronological records are intermediate between the two.

Fire frequency of the boreal forest of Alaska is highly variable at the ecoregion scale over short temporal scales but this variability is consistent over longer time scales. Over short time periods, landscape-level variations of estimated fire cycles range from 75 through 600 years (Gabriel and Tande 1983; Kasischke et al. 2002, 2010). For medium (centennial) to long time scales the same short term variability is present but the *mFRI* has been centered at 150-200 years (Anderson et al. 2006; Higuera et al. 2009; Lynch et al. 2002, 2004; Tinner et al. 2006) since *ca.* 5500 BP in interior Alaska and *ca.* 2400 BP in boreal ecoregions south of the Alaska Range (Hu et al. 2006). The North American boreal forest fire regime is characterized by large climate-driven variations of interannual and interdecadal area burned (Duffy et al. 2005; Hess et al. 2001; Macias Fauria and Johnson 2006, 2008; Skinner et al. 1999; Xiao and Zhuang 2007). Thus, regional variation in fire cycle may not represent spatially stratified fire regimes, and with the exception of treeline effects (Kasischke et al. 2002), controls of the physical environment are secondary to climate (Bessie and Johnson 1995). This interaction is reinforced by studies in boreal Alaska that fail to show statistical spatial differences in fire cycle or *FRI* (De Volder 1999, Drury and Grissom 2008).

Regional Differences

Despite the influence of short-term climate fluctuations there is still local variability of fire frequency in boreal Alaska (see figure 10). One variation is created by the Alaska Range, which separates the boreal forest of interior Alaska from smaller areas of boreal forest to the south. Interior Alaska, with its continental climate, has a higher lightning density than the Cook Inlet and Copper Plateau (Reap 1991) and may contribute to a shorter *mFRI* in interior Alaska. An analysis of fire records from 1959-79 supports this (Gabriel and Tande 1983). This study reports a fire cycle of 164 years for interior Alaska but over 10 000 years for boreal forests south of the Alaska Range.

However, the very long fire cycle estimated for areas south of the Alaska Range is likely to be an overestimate given the data only represent 20 years of fire history; too short to accurately assess fire cycle for a relatively small area, given the high interdecadal variability of area burned. Additionally, the data also represent the period of active fire suppression in Alaska, especially on the Cook Inlet where most of Alaska's population resides and fire suppression would have been most effective. Evidence of this is borne out by medium- to long-term paleoecological and dendrochronological studies in this area, which report a *mFRI* of150-210 years for the Copper River Plateau (Lynch et al. 2004) and 89 years for the Cook Inlet (De Volder 1999). However, paleoecological evidence does suggest that interior Alaska has a higher frequency of fires than boreal ecoregions south of the Alaska Range (Lynch et al. 2002), although the difference is less than that reported by Gabriel and Tande (1983). Lynch et al. (2003) also identify lightning frequency as the source of increased fire frequency in interior Alaska, since the frequency of fire at all sites increased in tandem during European settlement in the mid to late 19th century.

Within interior Alaska there is no evidence to suggest that fire frequency changes from south to north but spatial data do show a trend from west to east. To the north and south, the boreal forest of interior Alaska has sharp boundaries created by the Brooks and Alaska Ranges, respectively. Along both borders, climate, vegetation types, and fire regimes all change rapidly from forested areas dominated by dry warm summers and a high frequency of fire to tundra with cool summers and almost no fire. The lack of a latitudinal gradient of fire frequency in interior Alaska is likely due to the uniformity of climate (Trigg 1971) and fuels across this region (Gallant et al. 1995). This is contrary to the remainder of the North American boreal forest where the transition to tundra occurs across flat terrain and is accompanied by decrease in fire frequency (Johnson and Rowe 1975, 1977; Johnson 1979; Payette et al. 1989; Timoney and Wein 1991).

In interior Alaska the landscape is relatively flat and gradual changes in climate and fuels have a measurable impact on fire frequency from west to east. Fire frequency increases (Kasischke et al. 2002) and this is at least partially attributed to corresponding gradients in temperature and a decreasing precipitation gradient. This is similar to another west to east regional increase in fire frequency in Quebec, east of James Bay (Parisien and Sirois 2003). However, this trend is opposite the continental trends of decreasing fire frequency from west to east (Cyr et al. 2007, Kasischke and Turetsky 2006, Stocks et al. 2002);

The boreal ecoregions south of the Alaska Range are too small for analysis of regional scale trends. However, boreal Alaska taken as a whole does indicate a higher fire frequency from south to north divided by the Alaska Range. This is opposite the trend observed across Canada and is likely due to maritime climate, and lower lightning density during the burn season in boreal forest south of the Alaska Range. In contrast, fire frequency has been observed to increase from north to south towards the southern boundary of boreal forest in Finland (Larjavaara et al. 2005), far eastern Russia (Kharuk et al. 2007), and in Canada (Johnson 1992a).

Altitude

Relationships between topography and fire frequency have been reported for interior Alaska, the Cook Inlet, and Copper Plateau. Within interior Alaska fire frequency is inversely correlated with elevation, except in lowlands (less than 200 m), where the trend reverses (Kasischke et al. 2002). Lower fire frequency in lowland stands has also been detected in boreal forests of Ontario (Suffling and Molin 1982). Data from other studies also suggest that fire frequency decreases with elevation in interior Alaska (Drury and Grissom 2008, Goldstein 1981) and on the Kenai Peninsula (De Volder 1999). Studies elsewhere in the boreal forest have shown a similar relationship in Canada (Scott et al. 1987, Tande 1979) and in Russia (Kharuk et al. 2007). In the Noatak River basin of northwest Alaska (a tundra dominated region to the northwest of the Brooks Range with an unusually active fire regime) there is also a similar relationship between elevation and fire frequency in tundra (Racine et al. 1985). In contrast, few studies in the boreal forest of Canada have not been able to find a link between elevation and fire frequency (Masters 1990).

Aspect

To date, one study has examined the relationship between aspect and fire frequency in the boreal forest of Alaska; an analysis of several decades of fire records showed that within interior Alaska, a higher percentage of total burned area occurred on north facing slopes (44 percent) than on south-facing slopes (34 percent) (Kasischke et al. 2002). Kasischke et al. (2002) suggested that fuels on north-facing slopes support more fire activity because black spruce occur predominantly on north facing slopes (Viereck et al. 1986) and have been shown to have a higher rate of burning (Cumming 2001b) and density of lightning strikes (Dissing and Verbyla 2003) than other common boreal forest types. No other studies examining the relationship between fire frequency and aspect agree with Kasischke et al. (2002). In the boreal forest of Quebec, south and west facing slopes had higher fire frequency than north and east facing slopes (Cyr et al. 2007), while in the montane forests of western Canada there was no relationship with aspect (Masters 1990, Tande 1979). Higher fire frequency on south facing slopes is generally attributed to the effect of increased solar radiation on fuel moisture (Kunkel 2001), although in the boreal forest of Sweden, higher incidence of fire on south facing slopes has been attributed to human causes (Zackrisson 1977).

Terrain Features

Topographic position also has an influence on fire regimes. Fire history studies of small areas near Fairbanks, Alaska, found that the *mFRI* of concave land features is lower than adjacent areas (Barney 1971, Fastie et al. 2002). Both studies were conducted in the Caribou-Poker Creek Research Watershed, and observed that forests in small concave depressions or swales appeared resistant to fire and had no sign of fire for at least 200 years, whereas the estimated *mFRI* for the surrounding forest ranged from 40-60 years (Quirk and Sykes 1971) to 100-150 years (Fastie et al. 2002). These forests, frequently referred to as stringer forests, are small, on the order of acres or less, and are dominated by white spruce or a combination of white and black spruce. Quirk and Sykes (1971) noted the stringer forests were located in swales below escarpments with diffuse springs; the combination of sheltered topography and a water source created conditions, such as higher soil moisture and lower soil temperature, that were less conducive to fire spread than the surrounding landscape. Lower fire frequency has been associated with concave topographic features in boreal forests at other locations, including Quebec (Payette et al. 1989), and northern Sweden (Zackrisson 1977). Fastie et al. (2002) point out that while stringer forests appear to be fire free this may not actually be the case. Since fire history methodology is not always able to detect low severity surface fires, stringer forests may burn as frequently as the surrounding forests but overstory mortality may be low to absent.

Water features can also influence fire regimes by acting as fire breaks. In the boreal forest of Alaska, there is inconclusive evidence regarding an inverse relationship between the density of water features and fire frequency. Higher density of water features was significantly correlated with lower frequency of fire across interior Alaska (Kasischke et al. 2002), but on the Kenai Peninsula no relationship was found between fire frequency and density of water features, although the author notes that results could have been confounded by additional variables such as differential ignition between the two landscapes (De Volder 1999). In general fire history studies agree with Kasischke et al (2002). An inverse relationship between fire frequency and density of water features has been revealed in Labrador (Foster 1983), Quebec (Bergeron 1991, Bergeron et al. 2001), Alberta (Larsen 1997), and Sweden (Hellberg et al. 2004). One other fire history study, conducted in Quebec, did not find a significant relationship between fire frequency and the density of water features but the author notes that the study only looked at large fires which tend to be weather driven, and not small fires, which tend to be influenced more by terrain features (Grenier et al. 2005).

Vegetation Type

Reviewing fire frequency by vegetation type is, in a sense, redundant. To an extent, vegetation type is a function of the abiotic properties of the landscape (Viereck et al. 1986). In that respect the variation in fire frequency described below may be the same as some of the variations described above. However, this section is included because many fire history studies stratify the landscape by dominant overstory species of vegetation communities rather than abiotic landscape features. There is also debate about how much control abiotic landscape features have over the fire regime relative to the vegetation they support. An added complexity is climate which can change the type of fire regime or vegetation type supported at any given location over time.

This sub-section will focus primarily on the proportional differences in fire frequency among vegetation types. More specific information regarding fire regimes in individual vegetation types can be found in the chapter 3.

A large-scale spatial analysis (Yarie 1981) of stand age provides the best data to date on differences in fire frequency among forest types. In this study, the ages of 371 stands were calculated for the 3.6 million ha Porcupine and Upper Yukon River drainages in eastern Alaska. Stands were stratified by the dominant canopy type (black spruce, white spruce, and hardwood) and parametric models of stand age distribution were used to calculate the fire cycle for each type. Results showed that hardwood stands have the highest fire frequency, followed by black spruce and white spruce. To date there have been no other fire history studies to stratify fire frequency by vegetation type with comparative spatial extent but other smaller studies and data from rigorous studies in the North American boreal forest have either largely supported this distribution of fire frequency among vegetation types or have been inconclusive.

A fire history of 27 stands in the same area of Alaska stratified fire intervals by vegetation type (black spruce and white spruce/trembling aspen) (Drury and Grissom 2008). The results showed no significant difference in fire frequency between the two vegetation types though the data suggested that the white spruce/trembling aspen category had a longer fire frequency than the white spruce category. Two fire history studies near Fairbanks, AK, presented evidence that upland stands of black spruce and hardwoods have a higher fire frequency than adjacent white spruce stringer forests (Fastie et al. 2002, Quirk and Sykes 1971), but white spruce stringer forests represent only a small portion of the total cover of white spruce in boreal Alaska. These studies do identify stringer forest as a unique fire regime that likely indicates a fire-refugia type in boreal forest, but they cannot be taken to show that white spruce, the dominant overstory tree in stringer forests, in general has a lower fire frequency than hardwoods and black spruce.

A study similar to Yarie's (1981) stratified a large study area (44 870 km^2) in northern Manitoba into four forest types, jack pine, trembling aspen, white spruce, and black spruce. Although jack pine does not occur in Alaska, the differences in fire cycle between the other three forest types (Larsen 1997) were similar to results presented by Yarie (1981).

A different view of forest types and their respective fire frequency in boreal Alaska was presented in a large-scale spatial analysis of fire records in interior Alaska (DeWilde and Chapin 2006). Vegetation types for interior Alaska were divided into tundra, mixedwood boreal forest, spruce, and spruce-lichen woodland. Fire frequency was highest and similar in the two closed-canopy forest types (mixed hardwood-spruce and boreal spruce), followed by spruce-lichen woodland, and finally tundra. This trend is similar to the results of several fire history studies in Canada that show a latitudinal decrease in fire frequency from south to north that follows the progression from closed canopy boreal forest (spruce and mixed hardwood-spruce) to tundra, with spruce-lichen woodland as a transitional zone. Though not technically part of the boreal forest, alpine tundra is widely distributed throughout interior Alaska, and based on predominant vegetation types identified in physiographic provinces (Wahrhaftig 1965), occupies 32 percent of its land area. Thus, it is important to understand fire frequency in this vegetation type in relation to the surrounding boreal forest.

Chapter 3: Fire Regimes of Major Vegetation Types

There are four major vegetation types in the boreal forest of Alaska, with innumerable variations of each, and a number of unique minor vegetation types. Fire history studies primarily provide data only for broad classifications based on the major overstory species, for this reason, discussion of fire regimes is restricted to broad vegetation classes and some minor vegetation types with unique fire regimes.

Black Spruce Forest

Black spruce forests are well known for their higher flammability than other vegetation types (figure 11). Dense crowns with high foliar resin content often extend down into the surface fuels: a continuous layer of rapidly drying feathermosses and a dense layer of low-growing ericaceous shrubs. The combination of continuous surface fuels, frequent ladder fuels, and flammable canopy fuels creates optimal conditions for high-intensity crown fires that can initiate across a wide gradient of weather.

Statewide, black spruce is the most common forest type and occupies 39-44 percent of the area in interior Alaska (Calef et al. 2005, Viereck et al. 1986). Among other areas of boreal forest, it is the dominant vegetation type across the Copper Plateau but is less widely distributed in the Cook Inlet ecoregion (Gallant et al. 1995).

Black spruce can be broadly divided into lowland and upland types. Both are found at nutrient-poor sites where soils are cold and poorly drained, and the permafrost layer is shallow. Consequently, productivity is lowest of all the boreal forest types and the forest floor layer is well developed, often 50 cm greater in depth (van Cleve and Viereck 1981, van Cleve et al. 1991, Viereck et al. 1986). The primary difference between upland and lowland stands is topographic position: upland sites occur on low grade slopes and slopes with north facing aspects while lowland sites occupy abandoned river terraces in broad valleys (van Cleve et al. 1986). Other less common black spruce types include spruce-lichen woodlands at higher elevations and productive black spruce stands occasionally found on south facing slopes. Spruce-lichen woodlands, which form a transitional forest type between boreal forest and tundra, tend to occur on glacial deposits near latitudinal treeline. These climatic and geologic characteristics are uncommon in Alaska because much of the northern part of the state was un-glaciated during the last glacial period and the transition between boreal forest and tundra is highly compressed by the Brooks Range (Christiansen 1988). For this reason spruce-lichen woodlands are only a small part of the total black spruce forest type in Alaska. A fuelbed mapping project showed that spruce-lichen woodlands were restricted to the highlands of interior Alaska and only occupied 5 percent of the area (DeWilde and Chapin 2006).

The major difference between upland and lowland black spruce in forest structure and composition is the presence of wet site species in lowland stands, including tamarack, sphagnum moss (*Sphagnum* L.), and cotton grass (*Eriophorum* L.) (Viereck et al. 1986). There is no direct evidence of differences in the fire regime between these two forest types, although lowland black spruce is often interspersed with wetlands (Viereck et al. 1986) which may reduced fire frequency relative to upland stands (Bergeron 1991, Bergeron et al. 2001, Foster 1983, Hellberg et al. 2004, Kasischke et al. 2002). The spruce-lichen woodland is structurally more distinctive. Tree density is low and stands appear as open park-like forest with an overstory generally dominated by black spruce but usually including a component of white spruce. On drier sites the understory cover is dominated by several species of fruticose lichens as well as sparse cover of low growing shrubs, primarily blueberry (*Vaccinium* L.), Labrador tea (*Ledum* L.), and black crowberry (*Empetrum nigrum* L.) (Viereck et al. 1992).

Regardless of potential differences in fire regime among these three black spruce forest types the forest type in general has a short-interval stand-replacement fire regime. Based on previous estimates the fire cycle in black spruce forests is approximately 75 years. However spatial variability can be high owing to the stochastic nature of large fires and annual-to-decadal scale variability in area burned.

The most extensive study conducted on fire frequency of black spruce in regards to spatial area sampled and temporal depth of the data reported a fire cycle between 36 and 100 years for black spruce, depending upon the interpretation of the data (Yarie 1981). Similarly, a study of black spruce forests on the Kenai Peninsula also calculated a fire cycle of 42 years and a *mFRI* of 89 years (De Volder 1999). No other studies were spatially extensive enough to report fire cycle for black spruce in Alaska but two have estimated *mFRI*. One, conducted near Fairbanks, estimated the *mFRI* for black spruce as 102 and 146 years for two watersheds (Fastie et al. 2002), another in the Yukon Flats National Wildlife Refuge in eastern interior Alaska determined the *mFRI* was 67 years for black spruce stands (Drury and Grissom 2008). One dendrochronological study conducted in a spruce-lichen woodland on the southern flanks of the Brooks Range documented a number of fires but fires were not stand replacing (Christiansen 1988). Based on fire dates provided in the paper the *mFRI* would be 57 years, but this is based on only two intervals. This is within the range of *mFRI*s reported for spruce-lichen woodlands in western Canada (Johnson 1979).

Paleoecological studies have reported *mFRI* values for black spruce that are longer than the range reported in dendrochronological studies. Two studies have recently calculated fire frequency for the contemporary fire regime at two sites represented by black spruce. A *mFRI* of 150 years was calculated for the period 2000 years BP to the present for an area near McCarthy Alaska in the Copper Plateau (Lynch et al. 2004) and *mFRI* values of 150, 151, and 168 years were calculated for the period 5600 years BP to the present for three locations on the southern flanks of the Brooks Range (Higuera et al. 2009).

Black spruce is primarily burned in large stand-replacing crown fires. A recent study found that fires greater than 400 ha consume 97 percent of the area for lands in interior Alaska classified under the CFFDRS C-2 (Boreal Spruce) fuel model, primarily occupied by closed canopy black spruce forest (DeWilde and Chapin 2006).

The black spruce forest type is highly resilient to fires. Fires commonly result in extensive aboveground mortality and thus the main components of severity are unrelated to percentage of surviving vegetation but rather impacts on the forest floor and the degree of total fuel consumption. Regeneration shifts from surviving rootstock to seed sources as severity increases (Dyrness and Norum 1983, Johnstone and Kasischke 2005). Low-severity fires favor re-establishment of species present in the understory prior to the burn: the ericaceous shrubs, false toadflax (*Geocaulon lividum* (Richardson) Fernald), and bluejoint (*Calamagrostis canadensis* (Michx.) P. Beauv.). Whereas high-severity support an influx of early seral species including: green-tongue liverwort (*Marchantia polymorpha* L.*)* and fireweed (*Chamerion angustifolium* (L.) Holub). Black spruce re-colonizes via seed following fire. Cones are semi-serotinous and seedfall will continue for several years after the tree has died (Viereck and Dyrness 1979). Seedlings remain inconspicuous for one or two decades and then begin to overtop low growing shrubs. It is unclear how burn severity influences black spruce regeneration. Short term studies have shown that establishment and initial survival of black spruce is comparatively greater on sites with high burn severity (Dyrness and Norum 1983, Sirois and Payette 1991, Zasada et al. 1983), but longer term studies suggest that there is either no difference (Zasada et al. 1987) or lower overall density and biomass relative to pre-fire stand characteristics (Johnstone and Kasischke 2005).

The most common type of regeneration in black spruce is self replacement (Dix and Swan 1971, Viereck et al. 1983), although following high-severity fires instances of black spruce converting to forest dominated by trembling aspen have been documented (Johnstone and Kasischke 2005, Kasischke et al. 2000). It is unclear if the transition to aspen will be permanent as there is generally an understory of shade tolerant black spruce that may ultimately overtake the shade-intolerant aspen in the absence of fire (Foote 1983, Viereck 1983).

White Spruce Forest

White spruce forests are less common than black spruce forest in boreal Alaska. They occupy about 10 percent of the area within interior Alaska (Calef et al. 2005). There are three main types of white spruce forest: riparian, upland, and tree-line; each occupies a distinct topographic position and has unique properties.

Riparian white spruce is one of the most productive forest types in the Alaskan boreal forest (Viereck and Schandelmeier 1980). This forest type occurs along valley floors where growing conditions are ameliorated by sediment deposition from frequent flooding and a deep permafrost layer is suppressed by the relatively warm river water. Overstory composition can range from pure white spruce to mixed white spruce and balsam poplar.

With the exception of stringer forests, upland white spruce occurs on south-facing slopes less than 400 m above sea level with warm well drained soils and a deep permafrost layer (van Cleve et al. 1991, Viereck et al. 1986). Stringer forests occur as small stands (less than 10 ha) in wet slope-side depressions surrounded by drier site conditions (Quirk and Sykes 1971). Upland white spruce is relatively productive, though less so than riparian forests. Forest composition of upland white spruce on south facing slopes can be pure white spruce or include a component of paper birch or trembling aspen. Stringer forests may be pure white spruce or include a component of black spruce or paper birch.

Tree-line stands occur in a transitional band between boreal forest and tundra throughout the boreal forest. Soils are cold and permafrost is shallow. These forests are occasionally referred to as forest-tundra and have widely spaced slow-growing trees. This forest type has low productivity. Forest composition is generally a mixture of white and black spruce.

In general, all three white spruce forest types are considered less flammable relative to the black spruce and hardwood forest types (Viereck and Schandelmeier 1980). The white spruce fire regime may be best characterized as variable with both short-interval low-intensity fires and long-interval stand-replacing fires. Based on previous estimates the fire cycle in white spruce forests is approximately 110 years, but as with black spruce, variability is high. This variability encompasses the influence from fires in adjacent vegetation types, unique fuel dynamics of white spruce forest types, annual-to-decadal scale variability in area burned.

Comparative studies examining differences in fire frequency between forest types have consistently found that white spruce forests have a lower frequency of fire than adjoining forest types. In eastern interior Alaska, the fire cycle and *mFRI* of white spruce were 113 years (Yarie 1981) and 82 years (Drury and Grissom 2008), respectively, and both were longer than estimates for other forest types. The same trend was identified in a comparative paleoecological study in the Copper Plateau (Lynch et al. 2004). A paleoecological fire-history study found that areas of the Kenai Peninsula dominated by white spruce have a *mFRI* of 400-600 years (Berg and Anderson 2006), substantially longer than the fire frequency in nearby areas dominated by black spruce (De Volder 1999). This evidence is also supported by comparative studies in the boreal forest of Alberta (Larsen 1997), Saskatchewan (Weir et al. 2000), and Quebec (Parisien and Sirois 2003).

Among white spruce forest types, no significant differences among fire regimes have been reported. However, research focused on specific white spruce forest types provides insight into possible differences. It is likely that upland stands of white spruce on well drained sites and riparian white spruce forest have higher frequency of fire than the other two white spruce types (stringer forest and tree-line forest). Lynch et al. (2004) found that mixedwood boreal forest dominated by white spruce and upland hardwoods had an *mFRI* of 210 years. In interior Alaska fire history studies of riparian white spruce have found that fire is an important component of the disturbance regime (Farjon and Bogaers 1985, Mann and Plug 1999, Mann et al. 1995), but occurs less frequently than in surrounding upland forests and is not necessarily stand-replacing (Mann and Plug 1999). Others have argued that fire is not a part of the disturbance regime in riparian white spruce of Alaska and stands are regenerated by fluvial processes (van Cleve et al. 1991). This claim has been supported by a number of studies across the boreal forest of Canada identifying riparian white spruce stands that are several hundred years old with little or no evidence of fire (Nanson and Beach 1977, Viereck 1970, Walker et al. 1986).

In stringer forests and tree-line forests the importance of fire is reduced. Two dendrochronology studies found that fire was absent from stringer forests in interior Alaska for over 200 years (Fastie et al. 2002, Quirk and Sykes 1971), although low-intensity surface fires may have occurred with higher frequency. Evidence of fire is absent in tree-line stands of white spruce in Alaska (Goldstein 1981) and across the boreal forest of Canada (Payette and Filion 1985, Scott et al. 1987). It appears that stringer forests and treeline forests are resistant to fire due to consistently higher soil moisture during the fire season, which maintains higher fine-fuel and foliar moisture (Quirk and Sykes 1971).

White spruce forests are generally thought to be poorly adapted to frequent fire, although they do have adaptive traits that favor variable fire regimes where there is a long interval between stand replacing fires. Characteristics of white spruce that make it poorly adapted to fire include long interval to maturity (Lutz 1956, Neinstaedt and Zasada 1990), short seed dispersal range (Viereck 1973, Youngblood and Max 1992), thin bark, and non-serotinous cones. Traits that have allowed white spruce to maintain itself in a variable fire regime include the ability to survive low intensity fires, a tendency to produce mast seed crops in response to warm spring time conditions, which generally coincide with large fire years (Zasada and Gregory 1969), and a higher shade tolerance than competing hardwoods (Neinstaedt and Zasada 1990). Work on regeneration has also shown that white spruce regenerates well on burned sites provided there is a seed source (Densmore et al. 1999).

Regeneration following stand-replacing fire can follow two routes: self replacement or delayed regeneration beneath a canopy of hardwoods (paper birch and trembling aspen in upland stands and balsam poplar in riparian stands). Previous publications have suggested the latter type is most common in Alaska (van Cleve and Viereck 1981). This is supported by evidence from stand inventories that show decadent hardwoods are often a component of white spruce stands (Foote 1983) and paleoecological studies that indicate post-fire succession from hardwoods to white spruce (Larsen and MacDonald 1998). Evidence from stand-history studies in the boreal forest of Canada also indicates that this type of regeneration occurs frequently (Bergeron 2000). Throughout the boreal forest of North America, white spruce establishment occurs continuously for about three decades following fire, whereas hardwoods tend to establish more quickly (Dix and Swan 1971, Peters et al. 2006, van Cleve and Viereck 1981, Walker et al. 1986).

Hardwood Forest

After black spruce forest, hardwood forest is the second most common forest type in interior Alaska occupying 24 percent of the area (Calef et al. 2005). In the two southern ecoregions, hardwoods are dominant throughout the Cook Inlet but are relatively sparse in the Copper Plateau. There are three main types of hardwood forest: balsam poplar, paper birch, and trembling aspen. In Alaska, each species occurs in pure stands or mixed with one of the two spruce species but different hardwood species generally do not occur in the same stand.

Balsam poplar is the only tree species in Alaska that occurs north of the Brooks Range, where it forms small stands along spring fed streams (Bockheim et al. 2003). Generally this tree occurs in riparian areas as pure stands or mixed with white spruce. Growth rates are rapid and stands are highly productive relative to other boreal forest types (van Cleve et al. 1991, van Cleve et al. 1983, Zasada and Phipps 1990). Soils tend to be relatively warm river deposited sediment with a deep permafrost layer (Zasada and Phipps 1990), although in northern areas balsam poplar has been observed growing in areas with shallow permafrost (Gill 1972). In general the stands are narrow and confined to riparian areas impacted by frequent flooding, but large stands several kilometers wide do occur on the Tanana, Yukon, and Kuskokwim Rivers (Dyrness et al. 1986).

Paper birch is distributed throughout the boreal forest of Alaska, primarily oat upland sites on east and west facing slopes less than 600 m in elevation with relatively warm, well drained wind deposited soils with a deep permafrost layer (Foote 1983).

Trembling aspen occupies an ecological niche similar to paper birch and is generally found on well drained upland sites or abandoned river terraces (Foote 1983, Lutz 1956). Soils tend to be warmer than sites with paper birch and are well-drained to excessively well-drained with deep permafrost layers (Johnstone and Kasischke 2005, Lutz 1956, Rowe et al. 1974).

Both paper birch and trembling aspen stands are average in productivity (van Cleve and Viereck 1981) and can occur as pure stands or mixed with an understory component of white or black spruce (Foote 1983, Lutz 1956, Viereck et al. 1986).

Along with stringer forests and tree-line white spruce, balsam poplar likely represents a fire refugia in the boreal forests of Alaska. This fire regime is best classified as long-interval with low intensity. From isolated stands on the northern flanks of the Brooks Range to stands in interior Alaska, fire is rare in stands of balsam poplar (Lutz 1956). Stand age investigations report stands older than 200 years, indicating that at a minimum, stand-replacing fires are not common (Edwards and Dunwiddie 1985, Krause et al. 1959). Paleoecological studies have also shown that fire was relatively rare during periods when balsam poplar was historically a dominant part of the landscape (Higuera et al. 2009, Hu et al. 2006). However, information from these studies does not reveal the history of low-intensity surface fires in balsam poplar stands. The absence of fire in balsam poplar is logical given that stand flammability of boreal hardwood fuel types (Hely et al. 2001) and riparian forests (van Cleve et al. 1991) is low. However, given that balsam poplar is surrounded by more flammable forest types, it is feasible that these forests are periodically subject to low intensity surface fires. There have been no fire history studies in

Alaska that focused on stands of balsam poplar, but a fire-history study in Alberta did reveal that balsam poplar had low-intensity surface fires with a measured fire return interval of 20-30 years (Rood et al. 2007).

Despite the apparent lack of fire in stands of balsam poplar, it has adaptations to fire that make it both resistant and resilient. Balsam poplar has a thick bark that allows it to survive low-intensity surface fires. Re-sprouting is prolific following disturbance and also produces large crops of wind-dispersed seed (Lutz 1956, Rood et al. 2007, Viereck 1970, Zasada and Phipps 1990). However, it is perhaps more likely that these adaptations evolved as a response to the primary disturbances it is exposed to: ice damage and frequent flooding (Yarie et al. 1998).

Little is known about how balsam poplar regenerates following fire in the Alaskan boreal forest but in general it is thought to be either self-replacing or a mid-successional species that precedes white spruce following disturbance (Lutz 1956).

The fire regime for upland hardwoods (i.e. paper birch and trembling aspen) is different than balsam poplar. This forest type has the highest-frequency fire in boreal Alaska and would best be defined as a short-interval, low-intensity surface fire regime. Based on previous estimates of fire frequency in upland hardwoods, the fire cycle is approximately 55 years. There is less variability associated with this number than the fire cycle in black and white spruce forest types. However, this may be due to a smaller number of studies.

Yarie (1981) reported a fire cycle for upland hardwoods between 26 and 60 years, depending upon the interpretation of data. This is the only comparative study in Alaska that isolated hardwoods and the results show they have a considerably higher frequency of fire than either black or white spruce. Additional evidence of a high fire frequency in upland hardwoods comes from Drury and Grissom (2008). They measured a mean stand age of trembling aspen stands that was lower than both white and black spruce stands, indicating a higher fire frequency but no fire intervals were found in trembling aspen stands. Other calculations of *mFRI* for upland hardwoods in interior Alaska range from 40 to 65 years (Mann and Plug 1999, Quirk and Sykes 1971). This is generally lower than *mFRI*s reported for white and black spruce.

That Alaska's boreal upland hardwoods support the highest frequency of fire is well documented. This fire regime is also well documented across the boreal forest of Canada. Comparatively high fire frequencies have been documented in the Yukon Territory, British Columbia (Johnstone and Chapin 2006), Alberta (Larsen 1997), and Saskatchewan (Weir and Johnson 1998, Weir et al. 2000). Given that upland hardwoods are widely seen as a dominant canopy species during an intermediate seral stage (Bergeron 2000, Dix and Swan 1971, Foote 1983, Kay 1997, Larsen and MacDonald 1998, Quirk and Sykes 1971) some have argued that their presence has unduly influenced fire-history studies based on stand age distributions (Bergeron 1991, Cyr et al. 2007). If true studies in Alaska that use stand age data to infer fire frequency may overestimate fire frequency for the hardwood forest type (i.e. Drury and Grissom 2008, Yarie 1981).

Other studies in Alaska have calculated actual fire return intervals for upland hardwoods that are in agreement with Yarie's (1981) fire-cycle calculation for upland hardwoods (Mann and Plug 1999, Quirk and Sykes 1971). Only one comparative study has found that upland hardwoods have a longer fire frequency (2000 years) than black spruce (200 years) or white spruce (588 years) (Cumming 2001b).

Upland hardwoods are more likely to be burned by human-caused fires (66 percent) than by lightning-caused fires (34 percent). Human-caused fires tend to be smaller than lightning-caused fires and they extend the fire season by about one month in the spring and fall (DeWilde and Chapin 2006). It has been suggested that upland hardwoods have a fire season that begins and ends earlier than other boreal forest types since the leaf litter is exposed to the drying effects of the sun during spring before the canopy leafs out (Van Wagner 1983).

Flammability of upland hardwoods is low due to shading of dead fine fuels during the height of the fire season, a large gap between canopy fuels and surface fuels, and a high percentage of live fuels with low resin content (Mann and Plug 1999). Low flammability has been documented with fire-behavior models (Hely et al. 2001) and field experiments (Alexander and Lanoville 2004). The combination of high fire frequency and lower fire behavior is thought to be associated with to lower severity in upland hardwoods (Mann and Plug 1999), although other studies have documented high burn severity on south-facing slopes occupied by hardwoods (Cyr et al. 2007).

Upland hardwoods are resilient to fire, although there have been instances where upland hardwoods appeared to have survived low-intensity fires (Fastie et al. 2002). Regeneration adaptations following fire include large crops of wind dispersed seed and aggressive root sprouting. The ability to regenerate from root sprouts following fire has been widely documented for trembling aspen (Johnstone and Chapin 2006, Viereck and Dyrness 1979, Wang 2003), although stands initiating from seed have also been documented (Romme et al. 1997). Paper birch can only resprout from existing stumps and new post-fire stands are likely initiated by seed.

Boreal Tundra

An additional vegetation type worth discussing is tundra. Though not technically part of the boreal forest, tundra is widely distributed throughout interior Alaska and occupies 23-32 percent of the landscape (Calef et al. 2005, Wahrhaftig 1965). It is important to review the fire regime of boreal tundra because it is enmeshed within the boreal forest and is thus part of the fire regime at the landscape scale. Large areas of uplands within central and eastern interior Alaska are above treeline and tundra is a dominant cover type in western interior Alaska. The altitudinal transition from forest to alpine tundra is abrupt and the longitudinal transition in western interior Alaska is gradual.

Tundra within interior Alaska has a unique fire regime. Its fire frequency falls between that of the surrounding boreal forest and other regions of Alaska where tundra is the dominant vegetation type (Gabriel and Tande 1983, Racine et al. 1985, Wein 1976). The estimated fire cycle for physiographic provinces in interior Alaska where tundra is the dominant vegetation type is 237 years, compared to a fire cycle of 164 years for physiographic provinces in interior Alaska where the dominant vegetation type is boreal forest (Gabriel and Tande 1983). The largest areas of tundra within interior Alaska occur in the western third of interior Alaska where a slow transition from boreal forest to tundra occurs from east to west. Other large-scale spatial fire-history studies have calculated similar fire cycles for tundra within interior Alaska (Kasischke et al. 2002, Racine et al. 1985).

Of the vegetation types within tundra, dwarf birch shrub tundra appears to have the most active fire regime (de Groot and Wein 1999, Higuera et al. 2009). Scattered trees in upland areas is also associated with greater flammability (Wein 1976), but riparian treeline forests tend to be resistant to tundra fires (Racine et al. 1985, Wein 1976).

Most fires in tundra are small and restricted to fuel types on drier sites that have continuous cover of fuels. However, as with boreal forests, most of the area is burned by large fires. In tundra, this trend is exaggerated to an extent where one or two large fires may be responsible for nearly all of the area burned within a region (Gabriel and Tande 1983, Jones et al. 2009).

The fire season in boreal tundra is similar to the boreal forest and extends from May through August (Wein 1976). Most fires occur during a short period from June through July (Racine et al. 1985). Fire ignitions are both human- and lightning-caused. DeWilde and Chapin (2006) noted that human-caused fires tended to be disproportionately located in mixedwood boreal forest and tundra and concentrated in the beginning of the fire season. In contrast Racine et al. (1985) noted that all but one fire recorded in the Noatak Valley in western interior Alaska were lightning-caused. On a longitudinal gradient, lightning density is positively correlated with forest cover in interior Alaska but on an altitudinal gradient lightning is concentrated at the tree-line and falls in density with loss of elevation (Dissing and Verbyla 2003). This suggests that while lightning density may be correlated with fire cycle, there are geographic boundaries where other factors such as fuel dynamics override the influence of lightning, if not we would expect alpine tundra to have some of the highest frequency of fire within boreal ecoregions.

Other Vegetation Types

Several unique vegetation types were not covered above. This review did not include them because they either were not a major vegetation type or there was no published information regarding their fire regime. Some of these vegetation types, such as wetlands, likely have a lower-frequency fire regime than surrounding areas, while others, including south-facing shrub and aspen steppe, may have a higher frequency of fire.

Chapter 4: Temporal Trends in Boreal Fire Regimes

Millennial Trends for the Late Quaternary Period

Paleoecological studies have been ongoing in Alaska for over 40 years and a robust millennial-scale late Quaternary history of vegetation, fire, and climate has been reconstructed for the region from a growing database of paleorecords (Anderson and Brubaker 1994, Anderson et al. 2003, Edwards and Barker 1994, Edwards et al. 2001, Hu et al. 2006). In contrast, a framework to describe the basic dynamics among vegetation, fire, and climate remains murky (Hu et al.2006). This is a consequence of oversimplistic interpretation of patterns among fire, weather, and climate (Carcaillet and Richard 2000), low sample size (Gavin et al. 2006), poor resolution of climate data (Hu et al. 2006), and a lack of standardized techniques for quantifying fire history with paleorecords (Hu et al. 2006) .

Climate changed dramatically during the late Quaternary Period, from a glacial maximum 20 000-18 000 BP to an interglacial period lasting from 14 000 BP to the present (COHMAP 1988, Kutzbach et al. 1998, Wright et al. 1993). These broad trends in climate have been attributed to changes in solar insolation governed by orbital eccentricity of the Earth, the Laurentide Ice Sheet, and atmospheric CO_2 (Bartlein et al. 1991, Broccoli and Manabe 1987, Hays et al. 1976). Global patterns in climate change and variability from regional and local controls have been extensively documented in Alaska (Edwards et al. 2001, Elias 2001, Hu and Shemesh 2003, Hu et al. 1998, Hu et al. 2003, Kaufman et al. 2004).

Millennial-scale patterns of late Quaternary climate in central and eastern Beringia (extending from the Bering Land Bridge eastward to the MacKenzie River and including all of Alaska north of the Alaska Range) were associated with trends in fire frequency and vegetation composition, and can be divided into four general periods.

Herb Tundra Period, less than 14 000 BP. Charcoal data indicates fire was nearly absent vegetation cover was sparse (Higuera et al. 2009), however fire was important in less widely distributed locations dominated by birch shrub tundra (Earle et al. 1996).

Vegetation throughout was predominantly tundra with sparse cover of herbs, grasses, and prostrate shrubs (Anderson and Brubaker 1994, Higuera et al. 2009, Hu et al. 1993, Macdonald 1987, Ritchie 1982, 1985, Tinner et al. 2006) and localized areas of tall birch shrub tundra were present near the Arctic coastline (Anderson and Brubaker 1994, Edwards and Barker 1994, Ritchie and Hare 1971).

GCM model simulations and proxy climate data show that the climate during this period was much colder and drier than present (Edwards et al. 2001, Kutzbach et al. 1998). The Last Glacial Maximum (LGM) was *ca.* 21 000 BP and from the LGM through the end of this period, central and eastern Beringia remained ice free but isolated from other terrestrial landscapes in North America by the Laurentide Ice Sheet (Bartlein et al. 1998). High-resolution GCM simulations indicate that climate was moderating, with slowly increasing summer and winter average temperatures and increasing annual precipitation.

Shrub Tundra Period, 14 000-10 500 BP. Charcoal data indicate that fire frequency increased (Anderson et al. 2006, Earle et al. 1996, Higuera et al. 2009) and two estimates place the *mFRI* at approximately 140 years (Anderson et al. 2006, Higuera et al. 2009). Results from one location in Alaska show that fire was absent during this period, although vegetation composition at this site suggested balsam poplar gallery forests were common and thus may reflect a separate localized fire regime (Hu et al. 1993) or may simply be reflective of a methodology that is unable to detect fire in non-forested fuel types (Earle et al. 1996).

Vegetation composition during this period had a high degree of instability (Ritchie 1982, 1985). The predominant vegetation was tall shrub tundra dominated by shrub birch (Anderson et al. 2006, Cwynar and Spear 1991, Earle et al. 1996, Higuera et al. 2009, Hu et al. 1996, Macdonald 1987, Tinner et al. 2006). However, the transition from herb-grass-low shrub tundra was not simultaneous over the region but rather was geographically staggered over several millennia. Analyses of large paleoecological datasets in central and eastern Beringia have shown that tall birch shrub tundra established during the late glacial zone to the west (Anderson and Brubaker 1994) and north (Ritchie 1985) and essentially spread to the south and east over time (Anderson and Brubaker 1994, Macdonald 1987). However, the decline of tall birch shrub tundra was centered on the period between 11 500-10 500 BP, thus birch shrub tundra dominance was much longer in northern and western locations. One anomaly was the northeastern corner of central and eastern Beringia where decline of birch shrub tundra occurred earlier: *ca.* 13 300 BP (Ritchie 1982, 1985).

Overlapping the shrub tundra and spruce woodland periods were notable spikes of *Populus* spp. pollen percentages that ranged from 1000-2000 yrs in length and occurred from 13 000-8500 BP (Anderson and Brubaker 1994, Earle et al. 1996, Hu et al. 1996, Tinner et al. 2006). This period is commonly referred to as the poplar period and the high percentages of *Populus* spp. pollen have been interpreted as evidence of widespread distribution of balsam poplar. Generally, it is believed balsam poplar colonized riparian sites and south facing slopes embedded on a landscape otherwise dominated by birch shrub tundra and then white spruce woodlands (Anderson and Brubaker 1994). Other studies have categorized the poplar period as a more distinct vegetation type where the landscape was dominated by deciduous forests of balsam poplar and possibly paper birch (Higuera et al. 2009).

Ongoing warming during the late herb tundra period intensified and in Alaska the Holocene Thermal Maximum (HTM), period of peak temperatures, occurred at this time (Kaufman et al. 2004). Many factors were contributing to a warmer climate during this period including peak summer insolation *ca.* 11 000 BP (Berger 1978, Kutzbach et al. 1998), the retreat of the Laurentide Ice Sheet from most of Western North America (Kutzbach et al. 1998), and a doubling of atmospheric concentrations of CO_2 (Kutzbach et al. 1998). GCM simulations indicate that during the HTM climate was warmer than present in summer, colder than present in winter, and precipitation was greater than present (Bartlein et al. 1998). Climate analogue data and oxygen isotope analysis from this region also shows rapidly warming climate during this period (Edwards and Barker 1994, Edwards et al. 2001, Ellis and Calkin 1984, Hu and Shemesh 2003) and generally warmer than present summer temperatures (Anderson and Brubaker 1994, Elias 2001, Hu et al. 1996). Although GCM simulations indicate higher precipitation, climate analogue data indicate that effective moisture fluctuated widely (Abbott et al. 2000, Hu et al. 1998) and was always less than present (Abbott et al. 2000, Edwards et al. 2000, Hu et al. 1996, Hu et al. 1998).

Spruce Woodland Period, 10 500-7000 BP. Charcoal data suggest a variety of fire regimes were present but overall the importance of fire for interior Alaska and the Copper Plateau was lowest during this period (Earle et al. 1996, Higuera et al. 2009, Hu et al. 1996, Lynch et al. 2002). On the Kenai Peninsula, the opposite trend was observed based on data from three lakes (Anderson et al. 2006). Charcoal data from one lake just north of the Alaska Range indicated that this period represented a continuum of slowly increasing fire activity from the herb tundra zone through the present (Hu et al. 1993), although as mentioned above, there are concerns that fire activity in non-forested fuel types was underestimated. The two studies that indicated relatively short *mFRI*s for the prior period diverged during the spruce woodland period. Higuera et al. (2009) observed the *mFRI* lengthen to 251 years while Anderson et al. (2006) noted a shortening to 77 years. These studies were located in different regions of Alaska, the south-central Brooks Range and the Kenai Peninsula, respectively, and differences in *mFRI* may reflect regional variability.

Vegetation composition during this period was primarily white spruce woodland (Anderson and Brubaker 1994, Anderson et al. 2006, Cwynar and Spear 1991, Hu et al. 1993, Lynch et al. 2002, Macdonald 1987, Ritchie 1985, Tinner et al. 2006), but there were a number of secondary and transitional vegetation types.

The poplar period persisted for 1000-2000 years into the spruce woodland period in the south-central Brooks Range (Higuera et al. 2009), the Kenai Peninsula (Anderson et al. 2006), and western interior Alaska (Anderson and Brubaker 1994, Hu et al. 1996). Macroscopic charcoal records indicate that there was little change in *mFRI* as balsam poplar declined (Anderson et al. 2006, Higuera et al. 2009, Hu et al. 1996).

Another general trend was increasing alder pollen percentages during the spruce woodland zone (Anderson and Brubaker 1994, Cwynar and Spear 1991, Edwards and Barker 1994, Lynch et al. 2002, Tinner et al. 2006), indicating the increased cover of alder thickets. During this period the invasion of white spruce was restricted to eastern portions of interior Alaska and adjacent western Canada, while the dominant vegetation type in western interior Alaska remained birch shrub tundra with an increasing component of alder (Anderson and Brubaker 1994).

Climate proxy data indicate a stabilization and then reversal of climatic trends observed during the shrub tundra period (Abbott et al. 2000, Anderson et al. 2001, Anderson and Brubaker 1994, Edwards and Barker 1994, Ellis and Calkin 1984, Hu et al. 1998, Lynch et al. 2004). Paleoclimate data shows that effective moisture during the spruce woodland period, though less than present, was gradually increasing and was greater than the previous period (Abbott et al. 2000, Edwards et al. 2001, Finney et al. 2000). Climate analogues also indicate that summer temperatures began to cool during the last half of the spruce woodland period (Anderson and Brubaker 1994, Hu et al. 1998). GCM simulations point to several factors responsible for these trends including greater than present, but decreasing, summer insolation, rapid melting and eventual disappearance of the Laurentide Ice Sheet, and stabilization of atmospheric CO_2 concentrations at pre-industrial levels (Kutzbach et al. 1998). Model simulations reinforce interpretations of climate analogue data suggesting that the regional climate was warmer and drier than present but becoming cooler and wetter over time (Bartlein et al. 1998).

Boreal Forest Period, 7000-0 BP. Broadly, this period represents the establishment and stabilization of the current boreal fire regime in Alaska. With one exception on the Kenai Peninsula (Tinner et al. 2006), fire frequency increased during this zone (Earle et al. 1996, Higuera et al. 2009, Hu et al. 1993, 1996, Lynch et al. 2002). There was a widespread increase in fire frequency noted ca. 5500 BP (Higuera et al. 2009, Lynch et al. 2002, Tinner et al. 2006). The *mFRI* calculated during this period included values of 127 and 145 years for interior Alaska (Higuera et al. 2009, Lynch et al. 2002), 150-261 years for the Copper Plateau (Lynch et al. 2004, Tinner et al. 2006), and 130 years for the Cook Inlet (Anderson et al. 2006). Relative to interior Alaska, the fire regime for the Copper Plateau appears more variable and the modern fire regime did not establish until more recently (Lynch et al. 2004, Lynch et al. 2002).

The primary trend in vegetation during the boreal forest period was the replacement of white spruce woodlands by black spruce forest. This trend occurred across eastern interior Alaska, the Copper Plateau, and adjacent western Canada between 7500-6500 BP (Anderson and Brubaker 1994, Earle et al. 1996, Edwards and Barker 1994, Hu et al. 1993, 1996, Tinner et al. 2006). Later transitions were observed *ca.*

5500 BP in central interior Alaska (Higuera et al. 2009, Lynch et al. 2002, Tinner et al. 2006) and *ca*. 4600 BP on the Kenai Peninsula (Anderson et al. 2006). Further west (i.e. Seward Peninsula) and east (i.e. Yukon Territories), where present day vegetation is dominated by tundra and transitional vegetation, boreal forests were never a dominant part of the late Quaternary vegetation history. In western interior Alaska, shrub tundra continued to dominate the landscape during the spruce woodland zone and during the boreal forest zone, white spruce woodlands became an important component of the landscape *ca*. 6800 BP (Anderson and Brubaker 1994). In western Canada spruce woodlands continued to be predominant vegetation types from the spruce woodland zone into the boreal forest zone (Macdonald 1987, Ritchie 1985), and in some cases decreased in coverage and were ultimately replaced with tundra (Cwynar and Spear 1991).

Climate during the boreal forest zone converged on present day values and were stable relative to climate during the HTM. Climate analogue data indicate that directional trends for summer temperature and moisture during the end of the spruce woodland zone continued into the boreal forest zone (Abbott et al. 2000, Anderson et al. 2001, Calkin 1988, Edwards et al. 2001, Ellis and Calkin 1984, Higuera et al. 2009, Hu et al. 1996, Hu et al. 1998, Lynch et al. 2002). Analysis of climate analogue data shows that this period occurs following the end of the HTM, *ca*. 9000 (±2000) BP (Kaufman et al. 2004). Broadly, atmospheric CO_2 concentrations stayed at pre-industrial levels and summer insolation continued to decline. GCM simulations based on these broad trends indicate that relative to the spruce woodland period, the climate cooled primarily during the summer, but also slightly during the winter months, and measures of climate converged on present values (Bartlein et al. 1998).

Environmental Dynamics of Late Quaternary Millennial Scale Fire Regimes

The periods described above provide a broad description of the major changes in fire regimes, vegetation, and climate over the past 18 000 years (figure 12). These descriptions are based on general trends that represent eastern and central Beringia but significant local divergence from these trends has been observed. For more site specific descriptions one must review individual publications, or see Appendix 1, which has short summaries of all published fire history studies conducted in Alaska.

As mentioned at the beginning of this section, information from paleoecological fire history studies has been integrated with other paleorecords to explain millennial-scale trends in vegetation, fire, and climate, but attempts to explain the relationships between these three factors have been confounded. Comprehensive reviews of paleorecords have produced conflicting conclusions regarding the degree to which fuels and other factors exert influence on the fire regime (Carcaillet et al. 2001, Hu et al. 2006). It is unclear whether these conclusions reflect actual differences in environmental dynamics (i.e. interactions between fire, climate, and vegetation) or are simply different interpretations of the same system.

Poor resolution of environmental dynamics is to be expected. As current models show (Rupp et al. 2002), these interactions are complex and there is even widespread disagreement regarding contributing factors in contemporary fire regimes (Bessie and Johnson 1995, Cumming 2001b) where far more extensive and detailed data are available.

Given the complexity of environmental dynamics, explanations of interactions are hindered by limitations inherent in paleoecological studies. These include low temporal resolution of paleoecological data (Hu et al. 2006), vague or qualitative measures of climate (Hu et al. 2006), delayed reaction times of some climate analogues (Anderson and Brubaker 1994, Prentice 1986), small sample sizes (Gavin et al.

2006, Lynch et al. 2004), differences in fire history signals between microscopic and macroscopic charcoal analysis techniques (Tinner et al. 2006), and limited fire regime data.

Over the past decade many of the concerns listed above have, to some extent, been addressed. Isotope analysis and mutual climate range (MCR) analysis have provided quantitative climate history data with lower delay times (Anderson et al. 2001, Elias 2000, 2001, Hu and Shemesh 2003, Hu et al. 1998, Hu et al. 2003). GCMs have been improved and provide higher resolution climate simulations (Kutzbach et al. 1998). Macroscopic charcoal from lake sediment cores can provide more precise data on fire occurrence (Hu et al. 2006). And an ever increasing amount of data can be pooled to provide more statistical power to paleoecological analyses (Power et al. 2010). While these improvements have shed more light on long term fire regime dynamics uncertainties still remain.

Millennial-scale changes in fire regimes in Alaska are controlled by vegetation type with secondary climatic influences at decadal and centennial scales (Higuera et al. 2009, Hu et al. 2006). Support for this comes from the association between vegetation transitions and attendant changes in fire frequency. The most widely discussed is the transition from white spruce woodlands to black spruce forest during the mid-Holocene. The change was spurred by substantial increases in effective moisture and cooler temperatures between 7000-4000 BP (Anderson et al. 2001, Edwards et al. 2001, Lynch et al. 2004) and several fire history studies show the *mFRI* shortened from greater than 200 years to 100-200 years during this period (Higuera et al. 2009, Lynch et al. 2002, Tinner et al. 2006). Increasing fire frequency was also noted in several qualitative assessments of fire regimes in Alaska (Earle et al. 1996, Hu et al. 1993, 1996). Given the cooler and wetter climate, fire frequency should have decreased; since it instead increased, the change has been widely attributed to vegetation, which became more flammable (Higuera et al. 2009, Lynch et al. 2002). Furthermore, Hu et al. (2006) note that the fire regime shift was more closely correlated with vegetation transitions, which were staggered, rather than climate shift, which was geographically uniform, thus indicating that the fire regime was responding to changes in vegetation rather than climate. Individual fire history studies in Alaska have also noted a tighter correlation between fire and vegetation than between fire and climate during the mid Holocene (Lynch et al. 2004, Lynch et al. 2002).

Other findings contradict this viewpoint. On the Kenai Peninsula, paleoecological evidence shows the opposite fire frequency trend as white spruce woodlands were replaced by black spruce forests during the mid-Holocene. In this case, observing the better fit between climate and fire, the authors attributed changing fire frequency to climate (Anderson et al. 2006).

To highlight the difficulty of using paleoclimate data to infer fire-climate relationships it is instructive to examine conclusions of studies conducted outside of eastern and central Beringia. Carcaillet et al. (2001) studied a large paleoecological dataset representing a wide swath of boreal forest in eastern Canada. Mid-Holocene patterns in vegetation, climate, and fire frequency were similar to those observed in Alaska, but in this case the shift in fire frequency was attributed to climate rather than vegetation. The reasoning for this assessment is rooted in interpretation of the seasonal weather patterns. In eastern Canada, winter precipitation, which has no bearing on the fire regime, is a major driver of lake level, frequently used as a climate analogue for effective moisture (Carcaillet and Richard 2000). Based on reconstructions of paleoclimate, Carcaillet et al. (2001) reasoned that in eastern Canada, the cooler and wetter climate following the shift to black spruce boreal forest in the mid-Holocene actually increased variability of summer moisture regimes and hence amplified periods of extended seasonal drought. Given this, they argued, long-term general measures of climate could not be interpolated to represent the fire season.

The conclusion reached by Carcaillet et al. (2001) may represent regional divergence in environmental dynamics rather than a contradiction of conclusions reached in similar systems in Alaska (Higuera et al. 2009, Hu et al. 2006). Most precipitation in Alaska falls during the growing season (Hess et al. 2001, Lynch et al. 2004), thus unlike eastern Canada, lake level history in Alaska could be an indicator for summer moisture regimes. Similarly, Rupp et al. (2000) noted that their state and transition disturbance dynamic model (ALFRESCO) developed for the boreal forest of Alaska was not able to correctly retrodict historic patterns of vegetation and fire in eastern Canada, indicating a different set of relationships between vegetation, fire, and climate.

Other climatic variables shown to influence fire regimes but not preserved in climate analogues include intra- and interseasonal precipitation variability and lightning frequency (Kasischke et al. 2002, Larsen 1996). In Alaska, these trends have been suggested as possible reasons for millennial-scale shifts in fire frequency when vegetation and reconstructed climate provided insufficient explanations (Lynch et al. 2004). Climate reconstructions that resolve these climatic variables are needed to elucidate the relationship between fire and climate.

There is even less evidence to describe vegetation-climate-fire interactions for earlier periods (Hu et al. 2006).

Paleoecological pollen and charcoal data indicate a strong increase in fire frequency after herb tundra was replaced by shrub tundra during late-glacial period (Higuera et al. 2009, Higuera et al. 2008). Higuera et al. (2008, 2009) attribute the shift in fire frequency to vegetation change and make the case that shrub tundra, dominated by dense thickets of tall resinous shrub birch, would have been more conducive to the spread of fire than the sparse cover of tundra that preceded it, but not climatic trends, which they believe were not conducive to increased fire frequency. However, some climate reconstructions (Elias 2000, Hu et al. 1996) and GCM simulations (Bartlein et al. 1998, Kutzbach et al. 1998) have indicated that the climate during the late glacial period was likely several degrees warmer and drier than present. This would indicate that climate may have been a driver of the fire regime. Indeed, other paleoecological studies have speculated that both climate and vegetation change played a role as the fire regime became more active during the late-glacial period (Tinner et al. 2006).

Three paleoecological fire history studies cover the transition to the poplar period *ca.* 10 500 BP and there is little consistency among their conclusions on the relationship between early-Holocene stands of balsam poplar and the fire regime. Higuera et al. (2009) found that *mFRI* lengthened significantly as birch shrub tundra was replaced by balsam poplar woodlands, Earle et al. (1996) noted no qualitative change, and Anderson et al. (2006) found a significant shortening of *mFRI*. Higuera et al. (2009) reasoned that a warming and drier climate during the transition should have increased fire frequency but since deciduous forests are generally assumed to have low flammability they overwhelmed any climatic influence and caused the observed decrease. Although this conclusion is supported by studies that have shown boreal hardwood forests are less prone to burning than other fuel types (Amiro et al. 2001, Cumming 2001b, Hely et al. 2001), an opposite effect is shown by several fire history studies that show boreal hardwood forests have a relatively high fire frequency (Fastie et al. 2002, Larsen 1997, Mann and Plug 1999, Weir et al. 2000, Yarie 1981). Lack of agreement in modern records is confounded because early-Holocene deciduous woodlands have no modern analogue (Higuera et al. 2009). The uniformity of climate and vegetation change and diverging observations of fire frequency indicate that local-scale environmental disturbances can have substantial impacts on fire frequency.

The transition from deciduous woodlands to white spruce woodlands also suggests a complex environmental disturbance dynamic that varies along local to regional scales. Unlike the transition from white spruce woodlands to black spruce boreal forest, the timing of vegetation shifts was not well correlated with observed changes in fire frequency. Observed shifts during the mid-Holocene occurred generally during the same time *ca*. 8500 BP (Earle et al. 1996, Higuera et al. 2009, Hu et al. 1996, Tinner et al. 2006). The synchrony would suggest a possible link with regional climate patterns but the fire regime shift was not unidirectional among all sites (Anderson et al. 2006, Earle et al. 1996, Higuera et al. 2009, Hu et al. 1996, Tinner et al. 2006). Vegetation transitions to white spruce woodlands generally occurred about 1000 years prior to the mid-Holocene fire regime shift at sites in interior Alaska and south of the Alaska Range but were delayed at sites on the margins of the boreal forest of interior Alaska where they generally were associated with the shift (Higuera et al. 2009, Hu et al. 1996). If the mid-Holocene fire regime shift was impacted by climate it was manifested in multiple ways throughout Alaska in some cases increasing fire frequency while in other cases decreasing it or not changing at all.

Centennial to Decadal Scale Trends in Late Holocene Fire Regimes

The Late Holocene

Centennial- to decadal-scale fire history studies have generally been limited to the late-Holocene, after contemporary boreal forest had become established. These higher resolution analyses generally suggest that within millennial-scale fire regimes, centennial and decadal scale changes in climate and localized site differences can affect fire frequency (Gavin et al. 2006, Higuera et al. 2009, Hu et al. 2006).

Both precipitation and temperature have been linked with fire at decadal scales (Duffy et al. 2005, Larsen 1996, Larsen and Macdonald 1995). As one would expect, general measures of precipitation are on average negatively correlated with fire frequency. For instance, at Dune Lake in interior Alaska, *mFRI* shortens substantially from 198 years to 97 years *ca.* 2400 BP. This shift occurred in the absence of any measurable change in vegetation and was attributed to an increase in precipitation suggested by local climate analogue data (Lynch et al. 2002). Another example was the drier conditions during the Little Ice Age (LIA) (1500-1800 AD), which were associated with higher fire frequency at Grizzly Lake in the Copper Plateau (Tinner et al. 2008).

In fact, the pattern of high fire frequency observed during the dry and cold LIA, followed by decreasing fire frequency, was widely observed in boreal forests across the northern hemisphere and wholly or in part attributed to a dry and cold climate that preceded a period of warmer temperatures and higher precipitation in the 20^{th} century (Bergeron and Archambault 1993, Bergeron et al. 2001, Drury and Grissom 2008, Engelmark et al. 1994, Grenier et al. 2005, Johnson et al. 1998, Larsen 1997, Lehtonen and Kolstrom 2000, Lesieur et al. 2002, Niklasson and Granstrom 2000, Weir et al. 2000).

However, not all data support the negative correlation between precipitation and fire frequency. At the centennial scale, macroscopic charcoal data from Moose and Chokasna Lakes in the Copper Plateau indicated that *mFRI* was shorter than 300 years during periods of relatively wet weather and longer than 500 years during periods of relatively dry weather (Lynch et al. 2004). This counterintuitive trend was attributed to changes in lightning frequency and the influence of seasonal moisture variability.

Paleorecords from Moose and Chokasna Lakes also highlighted the effect of localized site differences on fire regimes. Moose Lake is located in mountainous terrain; Chokasna Lake is located within a lowland site with poor drainage (Lynch et al. 2004). The fire history record from Chokasna Lake showed a distinct shortening of *mFRI ca.* 2000 BP from 295 years to 150 years that was associated with a transition from mixed boreal forest to black spruce dominated boreal forest, but no change in either vegetation or fire frequency was observed at Moose Lake located *c.* 20 km away where the *mFRI* remained unchanged at 210 years. This difference was attributed to autogenic soil paludification at the Chokasna Lake site, which initiated succession from lower flammability mixed boreal forest to higher flammability black spruce forest.

European Settlement

Settlement coincided with increased fire frequency in interior Alaska (Duffy 2006, Fastie et al. 2002, Lynch et al. 2002) and on the Kenai Peninsula (De Volder 1999, Lynch et al. 2002). However, there are difficulties in determining the relative contribution of human activities since the climate was also changing during this period. Both De Volder (1999) and Lynch et al. (2002) admit it is not possible to determine the relative influences of European settlement activities on fire activity on the Kenai Peninsula. However a comparative study of stand age distributions in interior Alaska by Duffy (2006) strongly suggests that mining activity at two locations in interior Alaska increased fire frequency during the early part of the 20th century. Data from one of these locations near Fairbanks, AK, correspond with a period of more frequent fire reported by Fastie et al. (2002) for the same area and time period. Localized effects of human activity on fire frequency are consistent with patterns observed across interior Alaska today (DeWilde and Chapin 2006, Gabriel and Tande 1983) and lack of settlement may be one reason that Lynch et al. (2002) did not observe a large increase in charcoal at sites in interior Alaska. These results are consistent with trends observed in the boreal forests of Canada and Europe where localized areas of higher fire frequency were attributed to periods of settlement (Niklasson and Granstrom 2000, Weir and Johnson 1998). In Canada, comparative studies have also found that the effects of European settlement are localized (Weir et al. 2000), which may explain why many other studies have found no change in the fire regime during the period of European settlement (Bergeron 1991, Bergeron et al. 2001, Brassard and Chen 2006, Johnson and Fryer 1987, Johnson and Larsen 1991, Larsen 1997).

Fire Suppression

Early researchers agreed that fire suppression policies had effectively decreased fire frequency throughout boreal forests of the northern hemisphere (Barney and Stocks 1983, Engelmark 1984, Suffling and Molin 1982, Tande 1979, Van Wagner 1988, Wein and Moore 1977, 1979, Zackrisson 1977), including interior Alaska (Barney 1971, Yarie 1981). However, many of these studies relied on short term datasets (less than 50 years) and few considered alternative factors (e.g. climate change) (Suffling and Molin 1982). As mentioned above, the general pattern for northern latitudes (i.e. warmer and wetter climate as the LIA ended) was less conducive to fire spread. Thus, there is a danger that studies incorrectly attributed decreasing fire frequency to suppression when the actual cause was increasing precipitation (Johnson et al. 1998). This is not to imply that fire suppression did not have an impact on fire frequency; subsequent fire history studies recognized the potential compounding influence of climate and fire suppression activities and have found some influence from fire suppression activities (Bergeron et al. 2004b, Larsen 1996, Lefort et al. 2003).

In Alaska, conclusions regarding the influence of fire suppression reported in recent fire studies have been contradictory (DeWilde and Chapin 2006, Drury and Grissom 2008). Differences in conclusions are not necessarily mutually exclusive but rather reflect the spatial and temporal variation of the many factors that exert control over fire frequency and difficulty of measuring the impact of a short term event on a fire regime with high variability and average fire return intervals that exceed the length of the event. Indeed, the latter point is well illustrated by a decade-long debate concerning the relative influences of climate, fire suppression, and fuels on the fire regime of Ontario (Bridge et al. 2005, Johnson et al. 2001, Martell and Sun 2008, Miyanishi and Johnson 2001, Ward et al. 2001).

Looking into each of the four studies examining the influence of fire suppression in Alaska elucidates the complexity of the issue. The two earliest studies attributed decreased area burned to fire suppression during the suppression period (Barney 1971, Yarie 1981). However, multiple linear regression analysis of annual area burned on teleconnection indices explained 79 percent of variability in area burned for most of this period (Duffy et al. 2005), so except for potentially minor influences, the decadal trends observed are unlikely to be due to suppression. In contrast, a third study based on dendrochronology by Drury and Grissom (2008) observed that *mFRI*s on the Yukon Flats National Wildlife Refuge did not change appreciably among the periods before, during, and after the suppression era for the refuge. However, they conceded the difficulty of detecting the influence of fire suppression policy that only lasted a few decades based on *mFRI* in an area with long fire intervals and high interannual variability. The most definitive study conducted to date compared 15 years of fire records (1986-2000) from land classified as Critical or Full fire management options (i.e. fires are aggressively suppressed) against land classified as Modified or Limited (i.e. fires are generally allowed to burn unimpeded) (DeWilde and Chapin 2006). Results indicated that density of large fires and total area burned was lower in Critical and Full lands even when the study area was corrected for fuel type (i.e. only examined statistics for one fuel type, boreal spruce). This result agrees with other direct assessments of fire suppression in boreal Canada (Cumming 2005). However, even these results must be taken with some degree of caution as the dataset only spans 15 years and high variability in estimates of fire frequency would be inherent (Miyanishi and Johnson 2001).

DeWilde and Chapin (2006) suggest that fire suppression has likely had an impact on fire frequency in boreal Alaska, but they do so for only 1986-2000, when fire suppression policy allowed for a substantial number of fires to burn. Before that, from 1939 through the 1980s, the policy was to suppress all fires. Todd and Jewkes (2006) and DeWilde and Chapin's (2006) results cannot be extrapolated to this period. In the last two decades of the 20th century the effect of suppression on the fire regime is elusive. While it is true the policy was to suppress all fires, this did not necessarily translate into more effective suppression because resources, which were focused on full and critical suppression zones after the 1980's, would have to cover a much larger area. In fact prior to the 1960's, when a number of technological improvements in firefighting, chief among them the addition of aircraft, enabled more rapid detection and deployment of resources to remote areas (Todd and Jewkes 2006), suppression tactics were likely less successful than in the last two decades of the 20th century, despite the more aggressive policy.

Even given the lack of quantitative data on the relative influence of suppression policies on the fire regime in Alaska, the results from DeWilde and Chapin (2006) that fire suppression likely has at least some impact on lands under Full and Critical land management option. Given the statistical backcasting results from Duffy et al. (2005), we know that about 79 percent of the decreasing area burned trend observed by Barney (1971) and Yarie (1981) is attributable to changes in climate rather than fire suppression. Thus, the impact of fire suppression on fire frequency in Alaska is minor compared to the northern contiguous U.S. where suppression policy lengthened the fire cycle by one or two orders of magnitude and was able to effectively eliminate the influence of fire given the right amount of resources and road access (Frelich and Reich 1995, McCune 1983).

The Impacts of Human-caused Climate Change on Fire Regimes in Boreal Alaska

Over the last 30 years, fire management records indicate that annual area burned in the boreal forest has increased substantially (Kasischke and Turetsky 2006, Soja et al. 2007), but this trend is weaker or non-existent at smaller geographic scales (Kasischke et al. 2002, Stocks et al. 2002). Annual area burned has been positively correlated with drought and warmer than average growing season temperatures, which are in turn induced by the elevated presence of blocking high-pressure ridges over boreal North America (Duffy et al. 2005, Xiao and Zhuang 2007). The frequency of blocking highs is governed by teleconnection indices such as the Pacific Decadal Oscillation (PDO), the El Niño Southern Oscillation (ENSO), and the Arctic Oscillation (AO), which may be influenced by climate change (Macias Fauria and Johnson 2008). Decadal-scale effects of climate change on boreal fire regimes in North America will likely be increased annual area burned in western Canada and Alaska, with decreases in eastern Canada (Brassard and Chen 2006, Flannigan et al. 2005, Stocks et al. 1998, Wotton and Flannigan 1993, Zhang and Chen 2007). Specifically, this means a shorter fire cycle, longer burn season, and increased severity. The longer term, centennial-scale implications are unclear, as these projections are based only on changes in climate and do not account for feedback loops that may develop from interactions among environmental factors, primarily vegetation, fire, and climate.

One source of feedback loops that may have secondary influences on the fire regime is vegetation change. Paleoecological fire history studies have suggested that vegetation can exert a strong influence over the fire regime that may counteract climate change (Higuera et al. 2009, Hu et al. 2006), but these changes could only be resolved at the millennial scale and it is unclear if these changes occurred more rapidly. There are indications that changing fire regimes in the boreal forest have already influenced vegetation (Goetz et al. 2007), and ecological models have suggested changes in climate and fire could induce a transition from spruce to deciduous dominated forest in boreal Alaska (Calef et al. 2005, Potter 2004, Rupp et al. 2000a).

These forecasted changes in vegetation are in agreement with our current understanding of fire and vegetation interactions. High-severity burns have been shown to convert forest from spruce to deciduous types (Johnstone and Kasischke 2005, Johnstone et al. 2010a, Johnstone et al. 2004, Kasischke et al. 2000), and shorter fire cycles maintain deciduous forest (Mann and Plug 1999, Weir and Johnson 1998), which generally would revert back to spruce at longer fire cycles (Larsen and MacDonald 1998, Strong 2009). High resolution, centennial scale paleoecological studies also suggest that during periods of warmer temperatures individual fires can cause forests to shift from conifer to deciduous types (Larsen and MacDonald 1998, Tinner et al. 2006).

Assuming a conversion to deciduous forests in Alaska, there could be numerous feedbacks to the fire regime. Although stand age analysis may underestimate the fire cycle (Bergeron 1991), fire history studies based on other methods also found shorter fire cycles in deciduous dominated forests in Alaska and western Canada (Larsen 1997, Mann and Plug 1999, Quirk and Sykes 1971). This indicates that fire frequency could remain high following a shift to deciduous forest types but although severity and intensity would likely decrease (Cumming 2001b, Hely et al. 2001). A further layer of complexity is the ratio of black spruce to deciduous forest. Ecological models suggest that high fire frequency in deciduous stands is maintained by fires in adjacent stands of boreal spruce. A recent model scenario shows that when black spruce is converted to deciduous forest, overall fire frequency of the landscape decreased concomitantly with black spruce cover (Rupp et al. 2000b). This may explain differences in paleorecords that show deciduous forest types during the early Holocene had different fire regimes despite similar vegetation (Anderson et al. 2006, Higuera et al. 2009).

Another vegetation shift that could have impacts on the fire regime of boreal Alaska is the replacement of tundra by boreal forest (Gabriel and Tande 1983, Racine et al. 1985), which makes up much of interior Alaska and has a shorter fire cycle. In western interior Alaska, spruce forest is advancing into tundra, and sedge-dominated tundra is changing to shrub (primarily alder) tundra (Hinzman et al. 2005). The effect of climate change on treeline is unclear as ecological models (Rupp et al. 2000a) and paleoecological data (Anderson and Brubaker 1994) have shown only minor fluctuations of treeline in response to climate change, while recent data shows that the treeline is currently acceding. The overall effect of treeline migration on the fire regime presents some difficulties. The increased cover of forest would almost certainly increase the fire frequency if tundra converted to black spruce but if the conversion was to white spruce the flammability of the landscape would not likely change considerably (Christiansen 1988, Goldstein 1981). While birch shrub tundra has been shown to support frequent fire (Anderson et al. 2006), increases in alder are not likely to increase landscape flammability (Earle et al. 1996, Lynch et al. 2002).

It is clear that the immediate effects of climate change will be shorter fire cycle, longer burn seasons, and higher severity. As vegetation responds to this change, continuing changes in climate feedbacks will further alter the fire regime. Anticipating these feedbacks is difficult given localized controls, which could override regional patterns, and general assumptions about the flammability of vegetation types. Despite these uncertainties, however, it is important to study fire-environment interactions to better understand the role of fire and anticipate changes in climate, vegetation, and fire in order to optimize adaptation strategies.

Chapter 5: Key Findings and Conclusions

The information detailed in this report reflects the well documented spatiotemporal patterns of fire in the boreal forests of Alaska and dynamic interactions between fire and the environment. Despite the relative simplicity of the boreal biome, however, fire regimes and interactions with ecological processes are still sufficiently complex that a rigorous framework to explain them is elusive.

There are three overarching themes that confound our ability to describe fire-environment interactions more definitively.

1. Scale. While a number of interactions between fire and environmental variables have been catalogued their application is confined because as one moves between scales, either in time or space, these relationships can change.
2. Variability. While variability is inherent to fire regimes in North America, it is a defining aspect of the Alaskan boreal fire regime, notably at interannual to decadal scales (Kasischke et al. 2006).
3. Synergy. Rarely do fire-environment interactions occur in a vacuum. Factors that influence patterns of fire frequently interact with each other. Thus, relative contributions of individual drivers of the fire regime are difficult to ascertain.

These themes add context to and limit research conclusions.

Fire and land-management decisions are based on relatively short time horizons and spatially constrained by management boundaries. Over short time periods fluctuations in climate and area burned may be dramatic and appear to signal the onset of more permanent shifts in the fire regime when they are in fact within the historical range of variability. Moreover, even centennial-scale boreal fire regimes do not necessarily operate at equilibrium (Barclay et al. 2006), and have been undergoing directional shifts for several centuries. This adds further stress to management plans that are not adaptive, and casts doubt on policy directives that emphasize the importance of stable reference conditions (FRCC 2008).

During the last century, the primary drivers of area burned in the North American boreal forest at regional and continental scales were interannual- to decadal-scale atmospheric teleconnections (Duffy et al. 2005, Girardin and Sauchyn 2008, Hess et al. 2001, Macias Fauria and Johnson 2006, 2008, Skinner et al. 1999). Positive phases of these teleconnections can increase the presence of persistent blocking highs which bring periods of warmer drier weather and generate large fire years where area burned is substantially above average (Girardin and Wotton 2009, Hess et al. 2001, Zhang and Chen 2007). Most of the area burned in boreal Alaska and Canada is consumed during these large fire years by a relatively small number of large fires (larger than 50,000 ha) (Kasischke and Turetsky 2006, Kasischke et al. 2002). Statistical models have verified the relative importance of climate via correlations between annual area burned and teleconnections (Duffy et al. 2005, Hess et al. 2001). Climate-driven large fire years introduce challenges for research and management. High interannual-to-decadal variability of area burned can cause sudden and substantial shifts in sub-regional landscape vegetation mosaics and stand age distributions. The ecological impacts of large burns can impact management goals for wildlife habitat on small- to medium-sized land-management units. This variability also masks the initial stages of directional fire regime change (Girardin and Wotton 2009), and to overrides the influence of spatial heterogeneity (Podur and Martell 2009). These properties create uncertainty in management because precision of reported fire frequencies is low, geographic properties such as vegetation and topography do not necessarily make good boundaries for fire regimes, and trends are hard to spot and easy to misdiagnose.

Knowledge of centennial-scale fire-environment interactions are more poorly defined than those at shorter scales (Hu et al. 2006). A relatively small number of paleoecological fire history studies have examined fire regimes at this scale. Among these, temporal changes in the fire regime were poorly correlated among sampling locations within, and across studies. There was also a low degree of confidence for explanations of shifts in fire frequency (Lynch et al. 2004, Lynch et al. 2002, Tinner et al. 2006, Tinner et al. 2008). There are three main reasons for these shortfalls. The first is sample size. A number of studies with two or more sites in close proximity noted poor correlations between climate and fire frequency; this was unexpected given that sampling locations were only within miles of each other. The authors concluded that a larger number of sampling locations was needed to make inferences about broad influences of climate on fire regimes and suggested that local-scale spatial characteristics may cause uneven reactions to climate among sites (Gavin et al. 2006, Gavin et al. 2007, Lynch et al. 2004). The second is climate data. The relative importance of high spatial and temporal resolution climate data on fire occurrence and area burned in the boreal forest at annual- and decadal-scales has been well established (Balshi et al. 2009, Beverly and Martell 2005, Duffy et al. 2005, Flannigan and Harrington 1988, Girardin and Wotton 2009, Hess et al. 2001, Larsen 1996, Nash and Johnson 1996). Based on linkages with post-LIA climate warming and associated decreases in fire frequency, high-resolution climate data and spatially extensive fire-history data appear to be important to describing centennial-scale fire-climate interactions (Bergeron and Archambault 1993, Flannigan et al. 1998, Girardin and Sauchyn 2008, Weir et al. 2000, Zhang and Chen 2007). Prior to the LIA, historic climate records are largely absent and proxy records only provide more generalized relative (rather than absolute) measures of climate. The third reason is that vegetation change is a feedback to fire regimes. Decadal-scale fluctuations of fire frequency do not appear have a measurable effect on vegetation in long-interval fire regimes of the boreal forest (Drury and Grissom 2008). Centennial-scale vegetation changes associated with fluctuations of climate and fire have been documented (Larsen and MacDonald 1998, Tinner et al. 2008), and may serve as potential feedbacks to the fire regime. However, little is understood about these potential feedback loops.

Fire-environment interactions at millennial scales show a decreased role of climate and increased role of vegetation. Three recent publications, including a literature review and results from a paleoecological fire history study with a relatively large number of sample sites, concluded that vegetation is the major control of fire regimes (Higuera et al. 2009, Higuera et al. 2008, Hu et al. 2006). Others ascertain that climate is still a major driver of fire frequency at the millennial scale (Anderson et al. 2006) or document cases in which both climate and vegetation shifts appear to have contributed to changes in the fire regime (Earle et al. 1996, Hu et al. 1996, Lynch et al. 2002, Tinner et al. 2006). Although these studies rely on a small number of locations, which as previously mentioned for centennial-scale trends, can be misleading; a meta-analysis of paleoecological fire history data from boreal Canada also concluded that climate is the major driver of fire regimes at millennial scales (Carcaillet et al. 2001). These conclusions are based on measures of climate with low temporal resolution, such as lake level data and aquatic plant fossil analysis, and there is a substantial degree of uncertainty when attributing either vegetation or climate to a fire regime shift. However, long-term modeling based on vegetation as a millennial-scale driver of fire regimes does reinforce the role of vegetation as a major driver of millennial-scale fire regimes in Alaska (Brubaker et al. 2009).

Spatial heterogeneity of fire regimes has been well documented in Alaska. Individual trends often exist at fine to broad scales (i.e. stand to regional/ecozone), act as overlapping layers with varying degrees of interaction, interact with the multi-scale temporal trends, and have different degrees of variability.

Although a number of relationships between spatial attributes and fire regimes have been documented in boreal Alaska the relative contributions of spatial vs. temporal controls remain largely unknown. A number of studies in boreal Canada have concluded that spatial heterogeneity plays little or no role in spatial patterns of fire, although results may not be broadly applicable (Bessie and Johnson 1995, Grenier et al. 2005, Johnson and Larsen 1991, Johnson et al. 1998, Masters 1990). Others have determined that large fires are driven primarily by climate while small fires are controlled by spatial processes (Bergeron and Dansereau 1993, Dansereau and Bergeron 1993). Yet others have found that spatial drivers outweigh the effect of short-term climate trends on severity patterns (Barrett et al. 2010) and play an important role in establishing fire regimes at fine to broad scales (Cyr et al. 2007).

There are three broad-scale trends of fire frequency in boreal Alaska. The best defined is a west-to-east increase in fire frequency (Kasischke et al. 2002) and lightning density (Dissing and Verbyla 2003, Reap 1991). The two others include a decrease in fire frequency (Gabriel and Tande 1983, Lynch et al. 2002) and lightning density (Reap 1991) in boreal ecoregions south of the Alaska Range relative to interior Alaska and a negative correlation between fire frequency and mean waterbreak distance (MWD) in interior Alaska (Kasischke et al. 2002).

Fine-scale, stand-level spatial drivers of fire include vegetation type, canopy cover, aspect, topographic position, and elevation (Fastie et al. 2002, Kasischke et al. 2002, Mann and Plug 1999, Quirk and Sykes 1971, Yarie 1981). These drivers vary over small areas but broad patterns may have a measurable impact on fire regimes at the ecoregion scale (Kasischke et al. 2002). However, large climate-driven fires may override geographic and topographic influences. Prior to the large fire years of 2004 and 2005 the Yukon-Tanana Uplands ecoregion had a longer fire cycle than most other ecoregions in interior Alaska and this was attributed to spatial attributes including higher elevations, high relative precipitation and lower forest cover (Kasischke et al. 2002). After 2005 a follow-up analysis showed that the fire cycle of this ecoregion had converged with surrounding areas due to the impact of large fires in the previous two fire seasons (Kasischke et al. 2010) and suggests that any cumulative impacts of spatial drivers at broad scales may be rapidly overcome by large climate-driven fires.

Although spatial drivers appear to be secondary to temporal controls within the North American boreal forest (Dansereau and Bergeron 1993, Finney 1995, Podur and Martell 2009), spatial drivers are the primary control of fire statewide. There is a sharp increase in fire activity across the transition from adjacent major land cover types (i.e. tundra, coastal rainforest, barren mountains) to boreal forest, caused by spatial variation in fuels and climate (Gabriel and Tande 1983, Kasischke et al. 2002).

Land use patterns also have an impact of fire regimes, but at local scales. Paleoecological and dendrochronological data indicate that European settlement coincided with increased fire frequency (De Volder 1999, Fastie et al. 2002, Lynch et al. 2002, Yalcin et al. 2006), but this impact was likely limited to populated areas (Duffy 2006, Lynch et al. 2002). More recently, within 40 km of population centers fire suppression has reduced total area burned while human ignitions have increased the number of fires and shifted the beginning of the fire season back by about one month (Calef et al. 2008, DeWilde and Chapin 2006).

The subject of fire effects is an important extension of fire regimes and describes the impact of fire on ecosystem dynamics. Fire frequency and severity are important determinants of fire effects in boreal forests. Spatially heterogeneous burn patterns create high habitat diversity (Burton et al. 2008), and are part of a dynamic equilibrium that maintains broad landscape patterns over long time periods. During periods of shifting climate, fires act as catalyzing agents that increase rates of directional vegetation change over short to medium scales. Recent studies have focused on fire effects caused by more frequent and severe fire forecast to occur in response to climate change. In interior Alaska and adjacent areas of Canada, elevated fire frequency (Johnstone and Chapin 2006, Kasischke et al. 2000, Mann and Plug 1999) and severity (Barrett et al. 2010, Johnstone and Kasischke 2005, Johnstone et al. 2010a, Kasischke and Turetsky 2006, Kasischke et al. 2000, Mann and Plug 1999, Verbyla and Lord 2008) initiate stand-level transitions from conifer to hardwood forests. Additional fire effects caused by changes in fire frequency and severity include shifts in regeneration strategies (Peters et al. 2006), vegetation composition (Lieffers et al. 1993, Mann and Plug 1999, Turetsky et al. 2010), soil properties (Lecomte et al. 2006b), nutrient availability (Dyrness and Norum 1983, Simard et al. 2007), carbon sequestration (Amiro et al. 2001, Camill et al. 2009, Potter 2004), site productivity (Dyrness and Norum 1983, Lecomte et al. 2006a, Zasada et al. 1987), animal populations (Joly et al. 2009, Maier et al. 2005), and patterns of succession (Brassard and Chen 2006, Lecomte et al. 2006a).

Fires appear to catalyze directional climate-induced shifts in landscape-scale vegetation during decadal- to centennial-scale climatic change (Barrett et al. 2010, Johnstone et al. 2010b). This relationship is intuitive since increased fire activity is linked with warmer temperatures, but fire also catalyzes vegetation change during shifts to colder climates at the same temporal scale. Fire history documents fire-induced shifts from boreal forest to tundra at the stand and regional scale during periods of cooling climate during the late Holocene in Quebec, Canada (Filion et al. 1991, Payette and Morneau 1993, Sirois and Payette 1991).

Initial interactions between fire and climate are likely to change over time as feedback loops develop between fire and new fuelbeds. Consideration of landscape-scale vegetation shifts provides an example of how negative feedbacks can counter the influence of climate. In the contemporary North American boreal fire regime, fire history studies indicate the deciduous forest type has the most frequent fire (Mann and Plug 1999, Weir et al. 2000, Yarie 1981). Given the increasing fire frequency and a shift towards deciduous forest types, it would be tempting to assume that more frequent fire would be maintained at longer time scales once deciduous forests become dominant. However, numerous fire behavior studies show that fires spread more slowly in this forest type than in surrounding conifer forests (Alexander 2010, Cumming 2001b, Hely et al. 2000, Larsen 1997, Quintillo et al. 1991), and statistical model simulations suggest that current short fire intervals in deciduous forest types are maintained only when high-flammability conifer forests occupy a majority of the landscape (Rupp et al. 2002). Statistical model simulations are further supported by millennial-scale trends that show infrequent fire in landscapes dominated by deciduous forest types despite relatively warm and dry climate (Higuera et al. 2009, Hu et al. 2006).

Despite a number of knowledge gaps for fire-environment interactions recent attempts to retrodict historical fire regimes and vegetation trends based on known data have been encouraging (Brubaker et al. 2009, Duffy et al. 2005), and suggest that continued work to understand ecosystem and disturbance dynamics will yield improvements to ecological models that forecast the impacts of climate change.

Northern latitudes will be disproportionately affected by climate change (IPCC 2007) and recent warming at northern latitudes has been linked with greenhouse gas emissions (Gillett et al. 2004). Ecological disturbance models for this region have consistently forecast that a warming climate will increase fire activity (Flannigan and Van Wagner 1991, McCoy and Burn 2005, Stocks et al. 1998, Tymstra et al. 2007, Wotton and Flannigan 1993, Wotton et al. 2010), and cause large-scale vegetation shifts from conifer to deciduous boreal forest (Calef et al. 2005, Potter 2004, Rupp et al. 2001) and from herb to shrub tundra (Euskirchen et al. 2009). These forecasts are consistent with recent analyses of regional- to continental-scale changes of fire regimes (Kasischke and Turetsky 2006, Soja et al. 2007) and stand-level changes of boreal forest (Johnstone and Kasischke 2005, Johnstone et al. 2010a). However, uncertainty remains within frameworks used to describe ecosystem and disturbance dynamics (Hinzman et al. 2005, Rupp et al. 2007). However, more work is needed to explain the effects of teleconnections, which are important decadal-scale drivers of fire (Macias Fauria and Johnson 2008).

There is a degree of uncertainty surrounding longer centennial responses of the fire regime to future climate change and potential feedback loops to climate and fire caused by shifting vegetation patterns. Short-term forecasts largely indicate increases in fire frequency, fire season length, fire severity, and fire size, but rely primarily on changes in fire weather severity. However, paleoecological records indicate that past boreal landscape-scale shifts in fuel dynamics reduced fire activity over centennial-to-millennial scales despite inferred higher severity fire weather conditions (Higuera et al. 2009).

For the tundra biome, warming temperature may have increased fire frequency in both the short and long term. Larger tundra fires have recently been documented in Alaska (Jones et al. 2009). More frequent fire and warmer temperatures will favor a transition from herb to shrub tundra, including birch shrub community types (de Groot and Wein 1999), which have a high fine-fuel load and have shorter fire intervals than herb tundra in the paleoecological record (Higuera et al. 2008).

In conclusion, fires are the dominant disturbance in the Alaskan boreal forest and much has been learned about the relationship between fire and other components of the environment in the decades since formal fire management began in the state. More information does not necessarily make policy and management decisions any easier (Pyne et al. 1996), but it can increase awareness in management and policy, where decision makers must contend with increasingly complex choices as land-use pressures and climate-change impacts mount. It is with this in mind that this report was written. We hope this review and adjoining appendix will provide a reference for those with a stake in managing fire in the boreal forests of Alaska.

English Equivalents

When you know:	Multiply by:	To find:
Millimeters (mm)	.0394	Inches
Centimeters (cm)	.394	Inches
Meters (m)	3.28	Feet
Kilometers (km)	.621	Miles
Hectares (ha)	2.47	Acres
Square kilometers (km^2)	.386	Square miles
Degrees Celsius (C)	1.8 ºC + 32	Degrees Fahrenheit
Kilograms per square meter (kg/m^2)	.205	Pounds per square foot
Kilojoules per kilogram (kJ/kg)	.430	British thermal units per pound
Kilowatts per meter (kW/m)	.289	British thermal units per second per foot

References

Abbott, M.B.; Finney, B.P.; Edwards, M.E.; Kelts, K.R. 2000. Lake-level reconstructions and paleohydrology of Birch Lake, Central Alaska, based on seismic reflection profiles and core transects. Quaternary Research. 53(2): 154-166.

Agee, J.K. 1993. Fire ecology of Pacific northwest forests. Washington, D.C.: Island Press. 493 p.

Alaska Interagency Coordination Center. 2005. 2005 Alaska Wildland Fire Statistics. [Publisher and place of publication unknown]. 25 p.

Alaska Interagency Coordination Center. 2006. 2006 Alaska Wildland Fire Statistics. [Publisher and place of publication unknown]. 19 p.

Alaska Wildland Fire Coordinating Group. 1998. Alaska Interagency Wildland Fire Management Plan. [Juneau, AK: State of Alaska, Division of Natural Resources]. 61 p.

Alexander, M.E. 2010. Surface fire spread potential in trembling aspen during summer in the boreal forest region of Canada. Forestry Chronicle. 86(2): 200-212.

Alexander, M.E.; Lanoville, R.A. 1989. Predicting fire behavior in the black spruce-lichen woodland fuel type of western and northern Canada. Edmonton, AB & Fort Smith, NT: Forestry Canada, Northern Forestry Center, and Northwest Territories Provincial Government, Department of Renewable Resources, Territorial Forest Fire Center. 15 p.

Alexander, M.E.; Lanoville, R.A. 2004. The International Crown Fire Modeling Experiment fuel treatment trials. In: Engstrom, R.T.; Galley, K.E.M.; de Groot, W.J., eds. The 22nd Tall Timbers fire ecology conference: fire in temperate, boreal, and montane ecosystems. Tallahassee, FL: Tall Timbers Research Station: 222.

Alexander, M.E.; Stocks, B.J.; Lawson, B.D. 1991. Fire behavior in black spruce-lichen woodland: the Porter Lake project. Information Report NOR-X-310. Forestry Canada, Northwest Region, Northern Forestry Centre. 44 p.

Allen, J.L.; Sorbel, B. 2008. Assessing the differenced normalized burn ratio's ability to map burn severity in the boreal forest and tundra ecosystems of Alaska's national parks. International Journal of Wildland Fire. 17(4): 463-475.

Amiro, B.D.; Stocks, B.J.; Alexander, M.E.; Flannigan, M.D.; Wotton, B.M. 2001. Fire, climate change, carbon and fuel management in the Canadian boreal forest. International Journal of Wildland Fire. 10(3-4): 405-413.

Anderson, L.; Abbott, M.B.; Finney, B.P. 2001. Holocene climate inferred from oxygen isotope ratios in lake sediments, central Brooks Range, Alaska. Quaternary Research. 55(3): 313-321.

Anderson, P.M.; Brubaker, L.B. 1994. Vegetation history of northcentral Alaska: a mapped summary of late-Quaternary pollen data. Quaternary Science Reviews. 13(1): 71-92.

Anderson, P.M.; Edwards, M.E.; Brubaker, L.B. 2003. Results and paleoclimate implications of 35 years of paleoecological research in Alaska. Developments in Quaternary Science. 1: 427-440.

Anderson, R.S.; Hallett, D.J.; Berg, E.; Jass, R.B.; Toney, J.L.; de Fontaine, C.S.; De Volder, A. 2006. Holocene development of boreal forests and fire regimes on the Kenai lowlands of Alaska. Holocene. 16(6): 791-803.

Andrews, P.L.; Bevins, C.D.; Seli, R.C. 2005. BehavePlus fire modeling system, version 3.0: user's guide. Gen. Tech. Rep. RMRS-GTR-106WWW Revised. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 134 p.

Baldocchi, D.; Kelliher, F.M.; Black, T.A.; Jarvis, P. 2000. Climate and vegetation controls on boreal zone energy exchange. Global Change Biology. 6(Supplement 1): 69-83.

Balshi, M.S.; McGuire, A.D.; Duffy, P.; Flannigan, M.; Walsh, J.; Melillo, J. 2009. Assessing the response of area burned to changing climate in western boreal North America using a multivariate adaptive regression splines (MARS) approach. Global Change Biology. 15(3): 578-600.

Barclay, H.J.; Li, C.; Hawkes, B.; Benson, L. 2006. Effects of fire size and frequency and habitat heterogeneity on forest age distribution. Ecological Modelling. 197(1-2): 207-220.

Barney, R.J. 1971. Wildfires in Alaska – some historical and projected effects and aspects. In: Slaughter, C.W.; Barney, R.J.; Hansen, G.M., eds. Fire in the northern environment. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experimentation Station: 51-59.

Barney, R.J.; Stocks, B.J. 1983. Fire frequencies during the suppression period. In: Wein, R.W.; MacLean, D.A., eds. The role of fire in northern circumpolar ecosystems. New York: John Wiley and Sons Ltd.: 45-62. Chapter 3.

Barrett, K.; Kasischke, E.S.; McGuire, A.D.; Turetsky, M.R.; Kane, E.S. 2010. Modeling fire severity in black spruce stands in the Alaskan boreal forest using spectral and non-spectral geospatial data. Remote Sensing of Environment. 114(7): 1494-1503.

Bartlein, P.J.; Anderson, K.H.; Anderson, P.M.; Edwards, M.E.; Mock, C.J.; Thompson, R.S.; Webb, T.; Whitlock, C. 1998. Paleoclimate simulations for North America over the past 21,000 years: features of the simulated climate and comparisons with paleoenvironmental data. Quaternary Science Reviews. 17(6-7): 549-585.

Bartlein, P.J.; Anderson, P.M.; Edwards, M.E.; McDowell, P.F. 1991. A framework for interpreting paleoclimatic variations in eastern Beringia. Quaternary International. 10-12: 73-83.

Beaty, R.M.; Taylor, A.H. 2001. Spatial and temporal variation of fire regimes in a mixed conifer forest landscape, Southern Cascades, California, USA. Journal of Biogeography. 28(8): 955-966.

Berg, E.E.; Anderson, R.S. 2006. Fire history of white and Lutz spruce forests on the Kenai Peninsula, Alaska, over the last two millennia as determined from soil charcoal. Forest Ecology and Management. 227(3): 275-283.

Berger, A. 1978. Long-term variations of daily insolation and Quaternary climatic changes. Journal of Atmospheric Science. 35(12): 2362-2367.

Berger, A.; Loutre, M.F. 1991. Insolation values for the climate of the last 10 million years. Quaternary Science Reviews. 10(4): 297-317.

Bergeron, Y. 1991. The influence of island and mainland lakeshore landscapes on boreal forest-fire regimes. Ecology. 72(6): 1980-1992.

Bergeron, Y. 1998. Consequences of climate changes on fire frequency and forest composition in the southwestern boreal forest of Quebec. Géographie physique et Quaternaire. 52(2): 167-173.

Bergeron, Y. 2000. Species and stand dynamics in the mixed woods of Quebec's southern boreal forest. Ecology. 81(6): 1500-1516.

Bergeron, Y.; Archambault, S. 1993. Decreasing frequency of forest fires in the southern boreal zone of Quebec and its relation to global warming since the end of the 'Little Ice Age'. Holocene. 36(3): 255-259.

Bergeron, Y.; Dansereau, P. 1993. Predicting the composition of Canadian southern boreal forest in different fire cycles. Journal of Vegetation Science. 4(6): 827-832.

Bergeron, Y.; Gauthier, S.; Flannigan, M.; Kafka, V. 2004a. Fire regimes at the transition between mixedwood and coniferous boreal forest in northwestern Quebec. Ecology. 85(7): 1916-1932.

Bergeron, Y.; Flannigan, M.; Gauthier, S.; Leduc, A.; Lefort, P. 2004b. Past, current and future fire frequency in the Canadian boreal forest: Implications for sustainable forest management. Ambio. 33(6): 356-360.

Bergeron, Y.; Gauthier, S.; Kafka, V.; Lefort, P.; Lesieur, D. 2001. Natural fire frequency for the eastern Canadian boreal forest: consequences for sustainable forestry. Canadian Journal of Forest Research. 31(3): 384-391.

Bergeron, Y.; Leduc, A.; Harvey, B.D.; Gauthier, S. 2002. Natural fire regime: a guide for sustainable management of the Canadian boreal forest. Silva Fennica. 36(1): 81-95.

Bernhardt, E.; Hollingsworth, T.N.; Chapin III, F.S.; Viereck, L.A. 2011. Fire severity mediates climate driven shifts in understory composition of black spruce stands in interior Alaska. Journal of Vegetation Science**.** 22: 32-44.

Bessie, W.C.; Johnson, E.A. 1995. The relative importance of fuels and weather on fire behavior in subalpine forests. Ecology. 76(3): 747-762.

Beverly, J.L.; Martell, D.L. 2005. Characterizing extreme fire and weather events in the boreal shield ecozone of Ontario. Agricultural and Forest Meteorology. 133(1-4): 5-16.

Biswas, A.K.; Jayaweera, K.O.L.F. 1976. NOAA-3 satellite observations of thunderstorms in Alaska. Monthly Weather Review. 104(3): 292-297.

Black, R.A.; Bliss, L.C. 1978. Recovery sequence of *Picea mariana-Vaccinium uliginosum* forests after burning near Inuvik, Northwest Territories, Canada. Canadian Journal of Botany. 56(17): 2020-2030.

Boby, L.A.; Schuur, E.A.G.; Mack, M.C.; Verbyla, D.; Johnstone, J.F. 2010. Quantifying fire severity, carbon, and nitrogen emissions in Alaska's boreal forest. Ecological Applications. 20(6): 1633-1647.

Bockheim, J.G.; O'Brien, J.D.; Munroe, J.S.; Hinkel, K.M. 2003. Factors affecting the distribution of *Populus balsamifera* on the north slope of Alaska, USA. Arctic Antarctic and Alpine Research. 35(3): 331-340.

Bond, W.J.; Keeley, J.E. 2005. Fire as a global 'herbivore': the ecology and evolution of flammable ecosystems. Trends in Ecology & Evolution. 20(7): 387-394.

Bourgeau-Chavez, L.L.; Alexander, M.E.; Stocks, B.J.; Kasischke, E.S. 2000. Distribution of forest ecosystems and the role of fire in the North American boreal region. In: Kasischke, E.S.; Stocks, B.J., eds. Fire, climate change, and carbon cycling in the boreal forest. New York: Springer-Verlag: 111-131.

Bourgeau-Chavez, L.L.; Garwood, G.; Riordan, K.; Cella, B.; Alden, S.; Kwart, M.; Murphy, K. 2007. Improving the prediction of wildfire potential in boreal Alaska with satellite imaging radar. Polar Record. 43(227): 321-330.

Brassard, B.W.; Chen, H.Y.H. 2006. Stand structural dynamics of North American boreal forests. Critical Reviews in Plant Sciences. 25(2): 115-137.

Bridge, S.R.J.; Miyanishi, K.; Johnson, E.A. 2005. A critical evaluation of fire suppression effects in the boreal forest of Ontario. Forest Science. 51(1): 41-50.

Broccoli, A.J.; Manabe, S. 1987. The influence of continental ice, atmospheric CO_2, and land albedo on the climate of the last glacial maximum. Climate Dynamics. 1(2): 87-99.

Brubaker, L.B.; Higuera, P.E.; Rupp, T.S.; Olson, M.A.; Anderson, P.M.; Hu, F.S. 2009. Linking sediment-charcoal records and ecological modeling to understand causes of fire-regime change in boreal forests. Ecology. 90(7): 1788-1801.

Burton, P.J.; Parisien, M.A.; Hicke, J.A.; Hall, R.J.; Freeburn, J.T. 2008. Large fires as agents of ecological diversity in the North American boreal forest. International Journal of Wildland Fire. 17(6): 754-767.

Byram, G.M. 1959. Combustion of forest fuels. In: Forest fire: control and use. Davis, P.K., ed. New York: McGraw-Hill: 61-89.

Calef, M.P.; McGuire, A.D.; Chapin III, F.S. 2008. Human influences on wildfire in Alaska from 1988 through 2005: an analysis of the spatial patterns of human impacts. Earth Interactions. 12: 1-17.

Calef, M.P.; McGuire, A.D.; Epstein, H.E.; Rupp, T.S.; Shugart, H.H. 2005. Analysis of vegetation distribution in interior Alaska and sensitivity to climate change using a logistic regression approach. Journal of Biogeography. 32(5): 863-878.

Calkin, P.E. 1988. Holocene glaciation of Alaska (and adjoining Yukon Territory, Canada). Quaternary Science Reviews. 7(2): 159-184.

Camill, P.; Barry, A.; Williams, E.; Andreassi, C.; Limmer, J.; Solick, D. 2009. Climate-vegetation-fire interactions and their impact on long-term carbon dynamics in a boreal peatland landscape in northern Manitoba, Canada. Journal of Geophysical Research. 114, G04017. DOI: 04010.01029/02009JG001071.

Carcaillet, C.; Bergeron, Y.; Richard, P.J.H.; Frechette, B.; Gauthier, S.; Prairie, Y.T. 2001. Change of fire frequency in the eastern Canadian boreal forests during the Holocene: Does vegetation composition or climate trigger the fire regime? Journal of Ecology. 89(6): 930-946.

Carcaillet, C.; Richard, P.J.H. 2000. Holocene changes in seasonal precipitation highlighted by fire incidence in eastern Canada. Climate Dynamics. 16(7): 549-559.

Chambers, S.D.; Chapin III, F.S. 2002. Fire effects on surface-atmosphere energy exchange in Alaskan black spruce ecosystems: implications for feedbacks to regional climate. Journal of Geophysical Research. 108, 8145. DOI: 8110.1029/2001JD000530.

Chapin III, F.S.; Eugster, W.; McFadden, J.P.; Lynch, A.H.; Walker, D.A. 2000. Summer differences among Arctic ecosystems in regional climate forcing. Journal of Climate. 13(12): 2002-2010.

Chapin III, F.S.; Trainor, S.F.; Huntington, O.; Lovecraft, A.L.; Zavaleta, E.; Natcher, D.C.; McGuire, A.D.; Nelson, J.L.; Ray, L.; Calef, M.; Fresco, N.; Huntington, H.; Rupp, T.S.; DeWilde, L.; Naylor, R. L. 2008. Increasing wildfire in Alaska's boreal forest: pathways to potential solutions of a wicked problem. Bioscience. 58(6): 531-540.

Chen, H.Y.H.; Vasiliauskas, S.; Kayahara, G.J.; Ilisson, T. 2009. Wildfire promotes broadleaves and species mixture in boreal forest. Forest Ecology and Management. 257(1): 343-350.

Christiansen, J.S. 1988. A spruce-lichen woodland in northern Alaska: post-fire regeneration and community dynamics. Seattle, WA: University of Washington. 94 p. M.S. thesis.

COHMAP. 1988. Climate change of the last 18,000 years: observations and model simulations. Science. 241: 1043-1052.

Collins, W.B.; Helm, D.J. 1997. Moose, *Alces alces*, habitat relative to riparian succession in the boreal forest, Susitna River, Alaska. Canadian Field-Naturalist. 111(4): 567-574.

Cronan, J.B.; Jandt, R. 2008. How succession affects fire behavior in boreal black spruce forests of Alaska. Alaska Technical Report 59. Anchorage, AK: U.S. Department of the Interior, Bureau of Land Management, Alaska Fire Service. 15 p.

Cruz, M.G.; Alexander, M.E.; Wakimoto, R.H. 2005. Development and testing of models for predicting crown fire rate of spread in conifer forest stands. Canadian Journal of Forest Research. 35: 1626-1639.

Cumming, S.G. 2001a. A parametric model of the fire-size distribution. Canadian Journal of Forest Research. 31: 1297-1303.

Cumming, S.G. 2001b. Forest type and wildfire in the Alberta boreal mixedwood: What do fires burn? Ecological Applications. 11(1): 97-110.

Cumming, S.G. 2005. Effective fire suppression in boreal forests. Canadian Journal of Forest Research. 35(4): 772-786.

Cwynar, L.C.; Spear, R.W. 1991. Reversion of forest to tundra in the central Yukon. Ecology. 72(1): 202-212.

Cyr, D.; Gauthier, S.; Bergeron, Y. 2007. Scale-dependent determinants of heterogeneity in fire frequency in a coniferous boreal forest of eastern Canada. Landscape Ecology. 22(9): 1325-1339.

Dansereau, P.R.; Bergeron, Y. 1993. Fire history in the southern boreal forest of northwestern Quebec. Canadian Journal of Forest Research. 23(1): 25-32.

de Groot, W.J.; Pritchard, J.M.; Lynham, T.J. 2009. Forest floor fuel consumption and carbon emissions in Canadian boreal forest fires. Canadian Journal of Forest Research. 39: 367-382.

de Groot, W.J.; Wein, R.W. 1999. *Betula glandulosa* Michx. response to burning and postfire growth temperature and implications of climate change. International Journal of Wildland Fire. 9(1): 51-64.

De Volder, A. 1999. Fire and climate history of lowland black spruce forests, Kenai National Wildlife Refuge, Alaska. Flagstaff, AZ: Northern Arizona University. 127 p. M.S. thesis.

DeBano, L.F.; Neary, D.G.; Ffolliott, P.F. 1998. Fire's effects on ecosystems. New York: John Wiley & Sons, Inc. 233 p.

DeLong, S.C.; Kessler, W.B. 2000. Ecological characteristics of mature forest remnants left by wildfire. Forest Ecology and Management. 131(1-3): 93-106.

DeLong, S.C.; Tanner, D. 1996. Managing the pattern of forest harvest: lessons from wildfire. Biodiversity and Conservation. 5(10): 1191-1205.

Densmore, R.V.; Juday, G.P.; Zasada, J.C. 1999. Regeneration alternatives for upland white spruce after burning and logging in interior Alaska. Canadian Journal of Forest Research. 29(4): 413-423.

DeWilde, L.; Chapin III, F.S. 2006. Human impacts on the fire regime of interior Alaska: interactions among fuels, ignition sources, and fire suppression. Ecosystems. 9(8): 1342-1353.

Dissing, D.; Verbyla, D.L. 2003. Spatial patterns of lightning strikes in interior Alaska and their relations to elevation and vegetation. Canadian Journal of Forest Research. 33(5): 770-782.

Dix, R.L.; Swan, J.M.A. 1971. Roles of disturbance and succession in upland forest at Candle Lake, Saskatchewan. Canadian Journal of Botany. 49(5): 657-676.

Drury, S.A.; Grissom, P.J. 2008. Fire history and fire management implications in the Yukon Flats National Wildlife Refuge, interior Alaska. Forest Ecology and Management. 256(3): 304-312.

Duffy, P.A. 2006. Interactions among climate, fire, and vegetation in the Alaskan boreal forest. Fairbanks, AK: University of Alaska. 143 p. Ph.D. dissertation.

Duffy, P.A.; Epting, J.; Graham, J.M.; Rupp, T.S.; McGuire, A.D. 2007. Analysis of Alaskan burn severity patterns using remotely sensed data. International Journal of Wildland Fire. 16(3): 277-284.

Duffy, P.A.; Walsh, J.E.; Graham, J.M.; Mann, D.H.; Rupp, T.S. 2005. Impacts of large-scale atmospheric-ocean variability on Alaskan fire season severity. Ecological Applications. 15(4): 1317-1330.

Dyrness, C.T.; Norum, R.A. 1983. The effects of experimental fires on black spruce forest floors in interior Alaska. Canadian Journal of Forest Research. 13(5): 879-893.

Dyrness, C.T.; Viereck, L.A.; van Cleve, K. 1986. Fire in taiga communities of interior Alaska. In: van Cleve, K.; Chapin III, F.S.; Flanagan, P.W.; Viereck, L.A.; Dyrness, C.T., eds. Forest ecosystems in the Alaskan taiga. New York: Springer-Verlag: 74-86. Chapter 5.

Earle, C.J.; Brubaker, L.B.; Anderson, P.M. 1996. Charcoal in northcentral Alaskan lake sediments: relationships to fire and late-Quaternary vegetation history. Review of Palaeobotany and Palynology. 92(1-2): 83-95.

Eberhart, K.E.; Woodard, P.M. 1987. Distribution of residual vegetation associated with large fires in Alberta. Canadian Journal of Forest Research. 17(10): 1207-1212.

Edwards, M.E.; Barker, E.D. 1994. Climate and vegetation in northeastern Alaska 18,000 yr B.P.-present. Palaeogeography, Palaeoclimatology, Palaeoecology. 109(2-4): 127-135.

Edwards, M.E.; Bigelow, N.H.; Finney, B.P.; Eisner, W.R. 2000. Records of aquatic pollen and sediment properties as indicators of late-Quaternary Alaskan lake levels. Journal of Paleolimnology. 24(1): 55-68.

Edwards, M.E.; Dunwiddie, P.W. 1985. Dendrochronological and palynological observations on *Populus balsamifera* in northern Alaska, USA. Arctic and Alpine Research. 17(3): 271-277.

Edwards, M.E.; Mock, C.J.; Finney, B.P.; Barber, V.A.; Bartlein, P.J. 2001. Potential analogues for paleoclimatic variations in eastern interior Alaska during the past 14,000 yr: atmospheric-circulation controls of regional temperature and moisture responses. Quaternary Science Reviews. 20(1-3): 189-202.

Elias, S.A. 2000. Late Pleistocene climates of Beringia, based on analysis of fossil beetles. Quaternary Research. 53(2): 229-235.

Elias, S.A. 2001. Mutual climatic range reconstructions of seasonal temperatures based on late Pleistocene fossil beetle assemblages in eastern Beringia. Quaternary Science Reviews. 20(1-3): 77-91.

Ellis, J.M.; Calkin, P.E. 1984. Chronology of Holocene glaciation, central Brooks Range, Alaska. Geological Society of America Bulletin. 95(8): 897-912.

Engelmark, O. 1984. Forest fires in the Muddus National Park (northern Sweden) during the past 600 years. Canadian Journal of Botany. 62(5): 893-898.

Engelmark, O.; Kullman, L.; Bergeron, Y. 1994. Fire and age structure of Scots pine and Norway spruce in northern Sweden during the past 700 years. New Phytologist. 126(1): 163-168.

Epting, J.; Verbyla, D. 2005. Landscape-level interactions of prefire vegetation, burn severity, and postfire vegetation over a 16-year period in interior Alaska. Canadian Journal of Forest Research. 35(6): 1367-1377.

Epting, J.; Verbyla, D.; Sorbel, B. 2005. Evaluation of remotely sensed indices for assessing burn severity in interior Alaska using Landsat TM and ETM+. Remote Sensing of Environment. 96(3-4): 328-339.

Euskirchen, E.S.; McGuire, A.D.; Chapin III, F.S.; Yi, S.; Thompson, C.C. 2009. Changes in vegetation in northern Alaska under scenarios of climate change, 2003-2100: implications for climate feedbacks. Ecological Applications. 19(4): 1022-1043.

Farjon, A.; Bogaers, P. 1985. Vegetation zonation and primary succession along the Porcupine River in interior Alaska. Phytocoenologia. 13: 465-504.

Fastie, C.L.; Lloyd, A.H.; Doak, P. 2002. Fire history and postfire forest development in an upland watershed of interior Alaska. Journal of Geophysical Research. 107, 8150. DOI: 10.1029/2001JD000570. [printed 108(D1), 2003].

Fenton, N.; Lecomte, N.; Légaré, S.; Bergeron, Y. 2005. Paludification in black spruce (*Picea mariana*) forests of eastern Canada: potential factors and management implications. Forest Ecology and Management. 213(1-3): 151-159.

Filion, L.; Saint-Laurent, D.; Desponts, M.; Payette, S. 1991. The late Holocene record of aeolian and fire activity in northern Québec, Canada. The Holocene. 1(3): 201-208.

Finney, B.; Edwards, M.; Abbott, M.; Barber, V.; Anderson, L.; Rohr, M. 2000. Holocene precipitation variability in interior Alaska [Abstract]. In: 30th Arctic Workshop. Boulder, CO: University of Colorado, Institute of Arctic & Alpine Research. [Pages unknown].

Finney, M.A. 1995. The missing tail and other considerations for the use of fire history models. International Journal of Wildland Fire. 5(4): 197-202.

Flannigan, M.; Campbell, I.; Wotton, M.; Carcaillet, C.; Richard, P.; Bergeron, Y. 2001. Future fire in Canada's boreal forest: paleoecology results and general circulation model – regional climate model simulations. Canadian Journal of Forest Research. 31(5): 854-864.

Flannigan, M.D.; Bergeron, Y.; Engelmark, O.; Wotton, B.M. 1998. Future wildfire in circumboreal forests in relation to global warming. Journal of Vegetation Science. 9(4): 469-476.

Flannigan, M.D.; Harrington, J.B. 1988. A study of the relation of meteorological variables to monthly provincial area burned by wildfire in Canada (1953-80). Journal of Applied Meteorology. 27(4): 441-452.

Flannigan, M.D.; Logan, K.A.; Amiro, B.D.; Skinner, W.R.; Stocks, B.J. 2005. Future area burned in Canada. Climatic Change. 72(1-2): 1-16.

Flannigan, M.D.; Van Wagner, C.E. 1991. Climate change and wildfire in Canada. Canadian Journal of Forest Research. 21(1): 66-72.

Fleming, M.D. 1997. A statewide vegetation map of Alaska using a phenological classification of AVHRR data [Environmental]. Occasional Paper No. 52. Boulder, CO: University of Colorado, Institute of Arctic and Alpine Research.

Fleming, M.D.; Chapin III, F.S.; Cramer, W.; Hufford, G.L.; Serreze, M.C. 2000. Geographic patterns and dynamics of Alaska climate interpolated from sparse station data. Global Change Biology. 6 (Supplement 1): 49-58.

Foote, M.J. 1983. Classification, description, and dynamics of plant communities after fire in the taiga of interior Alaska. Res. Pap. PNW-307. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station. 109 p.

Foster, D.R. 1985. Vegetation development following fire in *Picea mariana* (black spruce)-*Pleurozium* forests of southeastern Labrador, Canada. Journal of Ecology. 73(2): 517-534.

Foster, D.R. 1983. The history and pattern of fire in the boreal forest of southeastern Labrador. Canadian Journal of Botany. 61(9): 2459-2471.

FRCC. 2008. Interagency fire regime condition class guidebook. 127 p.

Frelich, L.E.; Reich, P.B. 1995. Spatial patterns and succession in a Minnesota southern-boreal forest. Ecological Monographs. 65(3): 325-346.

French, N.H.F.; Kasischke, E.S.; Hall, R.J.; Murphy, K.A.; Verbyla, D.L.; Hoy, E.E.; Allen, J.L. 2008. Using Landsat data to assess fire and burn severity in the North American boreal forest region: an overview and summary of results. International Journal of Wildland Fire. 17(4): 443-462.

Gabriel, H.W.; Tande, G.F. 1983. A regional approach to fire history in Alaska. Technical Report 9. Anchorage, AK: U.S. Department of the Interior, Bureau of Land Management, Alaska State Office. 34 p.

Gallant, A.L.; Binnian, E.F.; Omernik, J.M.; Shasby, M.B. 1995. Ecoregions of Alaska. U.S. Geological Survey Professional Paper 1567. Washington, DC: United States Government Printing Office. 73 p.

Gauthier, S.; Bergeron, Y.; Simon, J.P. 1996. Effects of fire regime on the serotiny level of jack pine. Journal of Ecology. 84(4): 539-548.

Gavin, D.G.; Hallett, D.J.; Hu, F.S.; Lertzman, K.P.; Prichard, S.J.; Brown, K.J.; Lynch, J.A.; Bartlein, P.; Peterson, D.L. 2007. Forest fire and climate change in western North America: insights from sediment charcoal records. Frontiers in Ecology and the Environment. 5(9): 499-506.

Gavin, D.G.; Hu, F.S.; Lertzman, K.; Corbett, P. 2006. Weak climatic control of stand-scale fire history during the late Holocene. Ecology. 87(7): 1722-1732.

Gill, D. 1972. Point bar environment in Mackenzie River delta. Canadian Journal of Earth Sciences. 9(11): 1382-1393.

Gillett, N.P.; Weaver, A.J.; Zwiers, F.W.; Flannigan, M.D. 2004. Detecting the effect of climate change on Canadian forest fires. Geophysical Research Letters. 31, L18211. DOI: 10.1029/2004GL020876.

Girardin, M.P.; Sauchyn, D. 2008. Three centuries of annual area burned variability in northwestern North America inferred from tree rings. Holocene. 18(2): 205-214.

Girardin, M.P.; Wotton, B.M. 2009. Summer moisture and wildfire risks across Canada. Journal of Applied Meteorology and Climatology. 48(3): 517-533.

Goetz, S.J.; Mack, M.C.; Gurney, K.R.; Randerson, J.T.; Houghton, R.A. 2007. Ecosystem responses to recent climate change and fire disturbance at northern high latitudes: observations and model results contrasting northern Eurasia and North America. Environmental Research Letters. 2, 045031. DOI: 10.1088/1748-9326/2/4/045031.

Goldstein, G.H. 1981. Ecophysiological and demographic studies of white spruce (*Picea glauca* (Moench)Voss) at treeline in the central Brooks Range of Alaska. Seattle, WA: University of Washington. 193 p. Ph.D. dissertation.

Greene, D.F.; Noël, J.; Bergeron, Y.; Rousseau, M.; Gauthier, S. 2004. Recruitment of *Picea mariana*, *Pinus banksiana*, and *Populus tremuloides* across a burn severity gradient following wildfire in the southern boreal forest of Quebec. Canadian Journal of Forest Research. 34(9): 1845-1857.

Grenier, D.J.; Bergeron, Y.; Kneeshaw, D.; Gauthier, S. 2005. Fire frequency for the transitional mixedwood forest of Timiskaming, Quebec, Canada. Canadian Journal of Forest Research. 35(3): 656-666.

Hall, R.J.; Freeburn, J.T.; de Groot, W.J.; Pritchard, J.M.; Lynham, T.J.; Landry, R. 2008. Remote sensing of burn severity: experience from western Canada boreal fires. International Journal of Wildland Fire. 17(4): 476-489.

Hansen, A.J.; Spies, T.A.; Swanson, F.J.; Ohmann, J.L. 1991. Conserving biodiversity in managed forests – lessons from natural forests. Bioscience. 41(6): 382-392.

Hansen, J.; Ruedy, R.; Sato, M.; Reynolds, R. 1996. Global surface air temperature in 1995: return to pre-Pinatubo level. Geophysical Research Letters. 23(13): 1665-1668.

Harden, J.W.; Manies, K.L.; Turetsky, M.R.; Neff, J.C. 2006. Effects of wildfire and permafrost on soil organic matter and soil climate in interior Alaska. Global Change Biology. 12(12): 2391-2403.

Hare, F.K.; Ritchie, J.C. 1972. Boreal bioclimates. Geographical Review. 62(3): 332-365.

Hays, J.D.; Imbrie, J.; Shackleton, N.J. 1976. Variations in Earth's orbit: pacemaker of the ice ages. Science. 194(4270): 1121-1132.

Hellberg, E.; Niklasson, M.; Granström, A. 2004. Influence of landscape structure on patterns of forest fires in boreal forest landscapes in Sweden. Canadian Journal of Forest Research. 34(2): 332-338.

Hély, C.; Bergeron, Y.; Flannigan, M.D. 2000. Effects of stand composition on fire hazard in mixed-wood Canadian boreal forest. Journal of Vegetation Science. 11(6): 813-824.

Hély, C.; Flannigan, M.; Bergeron, Y.; McRae, D. 2001. Role of vegetation and weather on fire behavior in the Canadian mixedwood boreal forest using two fire behavior prediction systems. Canadian Journal of Forest Research. 31(3): 430-441.

Hess, J.C.; Scott, C.A.; Hufford, G.L.; Fleming, M.D. 2001. El Niño and its impact on fire weather conditions in Alaska. International Journal of Wildland Fire. 10(1): 1-13.

Heyerdahl, E.K.; Lertzman, K.; Karpuk, S. 2007. Local-scale controls of a low-severity fire regime (1750-1950), southern British Columbia, Canada. Ecoscience. 14(1): 40-47.

Higuera, P.E.; Brubaker, L.B.; Anderson, P.M.; Brown, T.A.; Kennedy, A.T.; Hu, F.S. 2008. Frequent fires in ancient shrub tundra: implications of paleorecords for Arctic environmental change. Plos One. 3(3), e0001744. DOI: 10.1371/journal.pone.0001744.

Higuera, P.E.; Brubaker, L.B.; Anderson, P.M.; Hu, F.S.; Brown, T.A. 2009. Vegetation mediated the impacts of postglacial climate change on fire regimes in the south-central Brooks Range, Alaska. Ecological Monographs. 79(2): 201-219.

Hillebrand, H. 2004. On the generality of the latitudinal diversity gradient. American Naturalist. 163(2): 192-211.

Hinzman, L.D.; Bettez, N.D.; Bolton, W.R.; Chapin III, F.S.; Dyurgerov, M.B.; Fastie, C.L.; Griffith, B.; Hollister, R.D.; Hope, A.; Huntington, H.P.; Jensen, A.M.; Jia, G.J.; Jorgenson, T.; Kane, D.L.; Klein, D.R.; Kofinas, G.; Lynch, A.H.; Lloyd, A.H.; McGuire, A.D.; Nelson, F.E.; Oechel, W.C.; Osterkamp, T.E.; Racine, C.H.; Romanovsky, V.E.; Stone, R.S.; Stow, D.A.; Sturm, M.; Tweedie, C.E.; Vourlitis, G.L.; Walker, M.D.; Walker, D.A.; Webber, P.J.; Welker, J.M.; Winker, K.; Yoshikawa, K. 2005. Evidence and implications of recent climate change in northern Alaska and other arctic regions. Climatic Change. 72(3): 251-298.

Hinzman, L.D.; Fukuda, M.; Sandberg, D.V.; Chapin III, F.S.; Dash, D. 2003. FROSTFIRE: an experimental approach to predicting the climate feedbacks from the changing boreal fire regime. Journal of Geophysical Research. 108, 8153. DOI: 8110.1029/2001JD000415.

Hoy, E.E.; French, N.H.F.; Turetsky, M.R.; Trigg, S.N.; Kasischke, E.S. 2008. Evaluating the potential of Landsat TM/ETM+ imagery for assessing fire severity in Alaskan black spruce forests. International Journal of Wildland Fire. 17(4): 500-514.

Hu, F.S.; Brubaker, L.B.; Gavin, D.G.; Higuera, P.E.; Lynch, J.A.; Rupp, T.S.; Tinner, W. 2006. How climate and vegetation influence the fire regime of the Alaskan boreal biome: the Holocene perspective. Mitigation and Adaptation Strategies for Global Change. 11: 829-846.

Hu, F.S.; Brubaker, L.B.; Anderson, P.M. 1993. A 12 000 year record of vegetation change and soil development from Wien Lake, Central Alaska. Canadian Journal of Botany. 71(9): 1133-1142.

Hu, F.S.; Brubaker, L.B.; Anderson, P.M. 1996. Boreal ecosystem development in the northwestern Alaska range since 11,000 yr B.P. Quaternary Research. 45(2): 188-201.

Hu, F.S.; Brubaker, L.B.; Gavin, D.G.; Higuera, P.E.; Lynch, J.A.; Rupp, T.S.; Tinner, W. 2006. How climate and vegetation influence the fire regime of the Alaskan boreal biome: the Holocene perspective. Mitigation and Adaptation Strategies for Global Change. 11: 829-846.

Hu, F.S.; Ito, E.; Brubaker, L.B.; Anderson, P.M. 1998. Ostracode geochemical record of Holocene climatic change and implications for vegetational response in the northwestern Alaska Range. Quaternary Research. 49(1): 86-95.

Hu, F.S.; Kaufman, D.; Yoneji, S.; Nelson, D.; Shemesh, A.; Huang, Y.; Tian, J.; Bond, G.; Clegg, B.; Brown, T. 2003. Cyclic variation and solar forcing of Holocene climate in the Alaskan subarctic. Science. 301(5641): 1890-1893.

Hu, F.S.; Shemesh, A. 2003. A biogenic-silica $\delta^{18}O$ record of climatic change during the last glacial-interglacial transition in southwestern Alaska. Quaternary Research. 59(3): 379-385.

Hutchison, K.O. 1968. Alaska's forest resources. Res. Bull. PNW-19. Juneau, AK: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station, Institute of Northern Forestry. 74 p.

IPCC. 2007. Climate change 2007: synthesis report. 73 p.

Jiang, Y.Y.; Zhuang, Q.L.; Flannigan, M.D.; Little, J.M. 2009. Characterization of wildfire regimes in Canadian boreal terrestrial ecosystems. International Journal of Wildland Fire. 18(8): 992-1002.

Johnson, E.A. 1979. Fire recurrence in the subarctic and its implications for vegetation composition. Canadian Journal of Botany. 57(12): 1374-1379.

Johnson, E.A. 1992a. Fire and vegetation dynamics: studies from the North American boreal forest. Cambridge: Cambridge University Press. 129 p.

Johnson, E.A. 1992b. Fire history and landscape pattern. Fire and vegetation dynamics: studies from the North American boreal forest. Cambridge: Cambridge University Press: 78-96. Chapter 6.

Johnson, E.A.; Fryer, G.I. 1987. Historical vegetation change in the Kananaskis Valley, Canadian Rockies. Canadian Journal of Botany. 65(5): 853-858.

Johnson, E.A.; Fryer, G.I.; Heathcott, M.J. 1990. The influence of man and climate on frequency of fire in the interior wet belt forest, British Columbia. Journal of Ecology. 78(2): 403-412.

Johnson, E.A.; Larsen, C.P.S. 1991. Climatically induced change in fire frequency in the southern Canadian Rockies. Ecology. 72(1): 194-201.

Johnson, E.A.; Miyanishi, K.; Bridge, S.R.J. 2001. Wildfire regime in the boreal forest and the idea of suppression and fuel buildup. Conservation Biology. 15(6): 1554-1557.

Johnson, E.A.; Miyanishi, K.; Weir, J.M.H. 1998. Wildfires in the western Canadian boreal forest: landscape patterns and ecosystem management. Journal of Vegetation Science. 9(4): 603-610.

Johnson, E.A.; Rowe, J.S. 1975. Fire in subarctic wintering ground of the Beverley caribou herd. American Midland Naturalist. 94(1): 1-14.

Johnson, E.A.; Rowe, J.S. 1977. Fire and vegetation change in the western subarctic. ALUR 75-76-61. Ottawa, ON: Department of Indian Affairs and Northern Development. 58 p.

Johnston, M.; Elliott, J. 1998. The effect of fire severity on ash, and plant and soil nutrient levels following experimental burning in a boreal mixedwood stand. Canadian Journal of Soil Science. 78(1): 35-44.

Johnstone, J. F.; Kasischke, E.S. 2005. Stand-level effects of soil burn severity on postfire regeneration in a recently burned black spruce forest. Canadian Journal of Forest Research. 35(9): 2151-2163.

Johnstone, J.F.; Chapin III, F.S. 2006. Fire interval effects on successional trajectory in boreal forests of northwest Canada. Ecosystems. 9(2): 268-277.

Johnstone, J.F.; Chapin III, F.S.; Foote, J.; Kemmett, S.; Price, K.; Viereck, L. 2004. Decadal observations of tree regeneration following fire in boreal forests. Canadian Journal of Forest Research. 34: 267-273.

Johnstone, J.F.; Hollingsworth, T.N.; Chapin III, F.S.; Mack, M.C. 2010a. Changes in fire regime break the legacy lock on successional trajectories in Alaskan boreal forest. Global Change Biology. 16(4): 1281-1295.

Johnstone, J.F.; Chapin III, F.S.; Hollingsworth, T.N.; Mack, M.C.; Romanovsky, V.; Turetsky, M. 2010b. Fire, climate change, and forest resilience in interior Alaska. Canadian Journal of Forest Research. 40(7): 1302-1312.

Johnstone, J.F.; Kasischke, E.S. 2005. Stand-level effects of soil burn severity on postfire regeneration in a recently burned black spruce forest. Canadian Journal of Forest Research. 35(9): 2151-2163.

Joly, K.; Jandt, R.R.; Klein, D.R. 2009. Decrease of lichens in Arctic ecosystems: the role of wildfire, caribou, reindeer, competition and climate in north-western Alaska. Polar Research. 28(3): 433-442.

Jones, B.M.; Kolden, C.A.; Jandt, R.; Abatzoglou, J.T.; Urban, F.; Arp, C.D. 2009. Fire behavior, weather, and burn severity of the 2007 Anaktuvuk River tundra fire, North Slope, Alaska. Arctic Antarctic and Alpine Research. 41(3): 309-316.

Kafka, V.; Gauthier, S.; Bergeron, Y. 2001. Fire impacts and crowning in the boreal forest: study of a large wildfire in western Quebec. International Journal of Wildland Fire. 10(2): 119-127.

Kane, E.S.; Kasischke, E.S.; Valentine, D.W.; Turetsky, M.R.; McGuire, A.D. 2007. Topographic influences on wildfire consumption of soil organic carbon in interior Alaska: implications for black carbon accumulation. Journal of Geophysical Research. 112, G03017. DOI: 03010.01029/02007JG000458.

Kasischke, E.S.; French, N.H.F.; O'Neill, K.P.; Richter, D.D.; Bourgeau-Chavez, L.; Harrell, P.A. 2000. Influence of fire on long-term patterns of succession in Alaskan boreal forest. In: Kasischke, E.S.; Stocks, B.J., eds. Fire, climate change, and carbon cycling in the boreal forest. New York: Springer-Verlag: 214-236. Chapter 12.

Kasischke, E.S.; Johnstone, J.F. 2005. Variation in postfire organic layer thickness in a black spruce forest complex in interior Alaska and its effects on soil temperature and moisture. Canadian Journal of Forest Research. 35(9): 2164-2177.

Kasischke, E.S.; O'Neill, K.P.; French, N.H.F.; Borgeau-Chavez, L.L. 2000. Controls on patterns of biomass burning in Alaskan boreal forests. In: Kasischke, E.S.; Stocks, B.J., eds. Fire, climate change, and carbon cycling in the boreal forest. New York: Springer-Verlag: 173-196. Chapter 10.

Kasischke, E.S.; Rupp, T.S.; Verbyla, D.L. 2006. Fire trends in the Alaskan boreal forest. In: Chapin III, F.S.; Oswood, M.S.; van Cleve, K.; Viereck, L.A.; Verbyla, D.L., eds. Alaska's changing boreal forest. Oxford: Oxford University Press: 285-301. Chapter 17.

Kasischke, E.S.; Turetsky, M.R. 2006. Recent changes in the fire regime across the North American boreal region – spatial and temporal patterns of burning across Canada and Alaska. Geophysical Research Letters. 33(13), L09703. DOI: 10.1029/2006GL025677.

Kasischke, E.S.; Turetsky, M.R.; Ottmar, R.D.; French, N.H.F.; Hoy, E.E.; Kane, E.S. 2008. Evaluation of the composite burn index for assessing fire severity in Alaskan black spruce forests. International Journal of Wildland Fire. 17(4): 515-526.

Kasischke, E.S.; Verbyla, D.L.; Rupp, T.S.; McGuire, A.D.; Murphy, K.A.; Jandt, R.; Barnes, J.L.; Hoy, E.E.; Duffy, P.A.; Calef, M.; Turetsky, M.R. 2010. Alaska's changing fire regime – implications for the vulnerability of its boreal forests. Canadian Journal of Forest Research. 40(7): 1313-1324.

Kasischke, E.S.; Williams, D.; Barry, D. 2002. Analysis of the patterns of large fires in the boreal forest region of Alaska. International Journal of Wildland Fire. 11(2): 131-144.

Kaufman, D.S.; Ager, T.A.; Anderson, N.J.; Anderson, P.M.; Andrews, J.T.; Bartlein, P.J.; Brubaker, L.B.; Coats, L.L.; Cwynar, L.C.; Duvall, M.L.; Dyke, A.S.; Edwards, M.E.; Eisner, W.R.; Gajewski, K.; Geirsdóttir, A.; Hu, F.S.; Jennings, A.E.; Kaplan, M.R.; Kerwin, M.W.; Lozhkin, A.V.; MacDonald, G.M.; Miller, G.H.; Mock, C.J.; Oswald, W.W.; Otto-Bliesner, B.L.; Porinchu, D.F.; Rühland, K.; Smol, J.P.; Steig, E.J.; Wolfe, B.B. 2004. Holocene thermal maximum in the western Arctic (0-180º W). Quaternary Science Reviews. 23(5-6): 529-560.

Kay, C.E. 1997. The condition and trend of aspen, *Populus tremuloides*, in Kootenay and Yoho National Parks: implications for ecological integrity. Canadian Field-Naturalist. 111(4): 607-616.

Key, C.H.; Benson, N.C. 2006. Landscape assessment: ground measure of severity, the composite burn index; and remote sensing of severity, the normalized burn ratio. In: Lutes, D.C; Keane, R.E.; Caratti, J.F.; Key, C.H.; Benson, N.C.; Sutherland, S.; Gangi, L.J., eds. FIREMON: Fire effects monitoring and inventory system. Gen. Tech. Rpt. RMRS-GTR-164-CD: LAI-15. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 51 p.

Kharuk, V.I.; Kasischke, E.S.; Yakubailik, O.E. 2007. The spatial and temporal distribution of fires on Sakhalin Island, Russia. International Journal of Wildland Fire. 16(5): 556-562.

Krause, H.H.; Rieger, S.; Wilde, S.A. 1959. Soils and forest growth on different aspects in the Tanana watershed of interior Alaska. Ecology. 40(3): 492-495.

Krebs, P.; Pezzatti, G.B.; Mazzoleni, S.; Talbot, L.M.; Conedera, M. 2010. Fire regime: history and definition of a key concept in disturbance ecology. Theory in Biosciences. 129(1): 53-69.

Kunkel, K.E. 2001. Surface energy budget and fuel moisture. In: Johnson, E.A.; Miyanishi, K., eds. Forest fires: behavior and ecological effects. San Francisco: Academic Press: 303-350. Chapter 13.

Kutzbach, J.; Gallimore, R.; Harrison, S.; Behling, P.; Selin, R.; Laarif, F. 1998. Climate and biome simulations for the past 21,000 years. Quaternary Science Reviews. 17(6-7): 473-506.

Larjavaara, M.; Kuuluvainen, T.; Rita, H. 2005. Spatial distribution of lightning-ignited forest fires in Finland. Forest Ecology and Management. 208(1-3): 177-188.

Larsen, C.P.S. 1996. Fire and climate dynamics in the boreal forest of northern Alberta, Canada, from AD 1850 to 1989. Holocene. 6(4): 449-456.

Larsen, C.P.S. 1997. Spatial and temporal variations in boreal forest fire frequency in northern Alberta. Journal of Biogeography. 24(5): 663-673.

Larsen, C.P.S.; Macdonald, G.M. 1995. Relations between tree-ring widths, climate, and annual area burned in the boreal forest of Alberta. Canadian Journal of Forest Research. 25(11): 1746-1755.

Larsen, C.P.S.; MacDonald, G.M. 1998. An 840-year record of fire and vegetation in a boreal white spruce forest. Ecology. 79(1): 106-118.

Lecomte, N.; Simard, M.; Bergeron, Y. 2006a. Effects of fire severity and initial tree composition on stand structural development in the coniferous boreal forest of northwestern Quebec, Canada. Ecoscience. 13(2): 152-163.

Lecomte, N.; Simard, M.; Fenton, N.; Bergeron, Y. 2006b. Fire severity and long-term ecosystem biomass dynamics in coniferous boreal forests of eastern Canada. Ecosystems. 9(8): 1215-1230.

Lefort, P.; Gauthier, S.; Bergeron, Y. 2003. The influence of fire weather and land use on the fire activity of the Lake Abitibi area, eastern Canada. Forest Science. 49(4): 509-521.

Lehtonen, H.; Kolström, T. 2000. Forest fire history in Viena Karelia, Russia. Scandinavian Journal of Forest Research. 15(6): 585-590.

Lentile, L.B.; Holden, Z.A.; Smith, A.M.S.; Falkowski, M.J.; Hudak, A.T.; Morgan, P.; Lewis, S.A.; Gessler, P.E.; Benson, N.C. 2006. Remote sensing techniques to assess active fire characteristics and post-fire effects. International Journal of Wildland Fire. 15(3): 319-345.

Lesieur, D.; Gauthier, S.; Bergeron, Y. 2002. Fire frequency and vegetation dynamics for the south-central boreal forest of Quebec, Canada. Canadian Journal of Forest Research. 32(11): 1996-2009.

Lieffers, V.I.; Macdonald, S.E.; Hogg, E.H. 1993. Ecology of and control strategies for *Calamagrostis canadensis* in boreal forest sites. Canadian Journal of Forest Research. 23(10): 2070-2077.

Lutz, H.J. 1956. Ecological effects of forest fires in the interior of Alaska. Technical Bulletin 1133. [Fairbanks, AK]: U.S. Department of Agriculture, Forest Service, Alaska Forest Research Center. 121 p.

Lynch, J.A.; Clark, J.S.; Bigelow, N.H.; Edwards, M.E.; Finney, B.P. 2002. Geographic and temporal variations in fire history in boreal ecosystems of Alaska. Journal of Geophysical Research. 108, 8152. DOI: 10.1029/2001JD000332.

Lynch, J.A.; Hollis, J.L.; Hu, F.S. 2004. Climatic and landscape controls of the boreal forest fire regime: Holocene records from Alaska. Journal of Ecology. 92(3): 477-489.

Macdonald, G.M. 1987. Postglacial vegetation history of the Mackenzie River basin. Quaternary Research. 28(2): 245-262.

Macias Fauria, M.; Johnson, E.A. 2006. Large-scale climatic patterns control large lightning fire occurrence in Canada and Alaska forest regions. Journal of Geophysical Research. 111, G04008. DOI: 10.1029/2006JG000181.

Macias Fauria, M.; Johnson, E.A. 2008. Climate and wildfires in the North American boreal forest. Philosophical Transactions of the Royal Society B. 363(1501): 2317-2329.

Maier, J.A.K.; Ver Hoef, J.M.; McGuire, A.D.; Bowyer, R.T.; Saperstein, L.; Maier, H.A. 2005. Distribution and density of moose in relation to landscape characteristics: effects of scale. Canadian Journal of Forest Research. 35(9): 2233-2243.

Mann, D.H.; Fastie, C.L.; Rowland, E.L.; Bigelow, N.H. 1995. Spruce succession, disturbance, and geomorphology on the Tanana River floodplain, Alaska. Ecoscience. 2(2): 184-199.

Mann, D.H.; Plug, L.J. 1999. Vegetation and soil development at an upland taiga site, Alaska. Ecoscience. 6(2): 272-285.

Martell, D.L.; Sun, H. 2008. The impact of fire suppression, vegetation, and weather on the area burned by lightning-caused forest fires in Ontario. Canadian Journal of Forest Research. 38(6): 1547-1563.

Masters, A.M. 1990. Changes in forest fire frequency in Kootenay National Park, Canadian Rockies. Canadian Journal of Botany. 68(8): 1763-1767.

McCoy, V.M.; Burn, C.R. 2005. Potential alteration by climate change of the forest-fire regime in the boreal forest of central Yukon Territory. Arctic. 58(3): 276-285.

McCune, B. 1983. Fire frequency reduced two orders of magnitude in the Bitterroot Canyons, Montana. Canadian Journal of Forest Research. 13(2): 212-218.

Miller, J.D.; Thode, A.E. 2007. Quantifying burn severity in a heterogeneous landscape with a relative version of the delta normalized burn ratio (dNBR). Remote Sensing of Environment. 109(1): 66-80.

Miyanishi, K.; Johnson, E.A. 2001. Comment – A re-examination of the effects of fire suppression in the boreal forest. Canadian Journal of Forest Research. 31(8): 1462-1466.

Miyanishi, K.; Johnson, E.A. 2002. Process and patterns of duff consumption in the mixedwood boreal forest. Canadian Journal of Forest Research. 32(7): 1285-1295.

Mock, C.J.; Bartlein, P.J.; Anderson, P.M. 1998. Atmospheric circulation patterns and spatial climatic variations in Beringia. International Journal of Climatology. 18(10): 1085-1104.

Morgan, P.; Hardy, C.C.; Swetnam, T.W.; Rollins, M.G.; Long, D.G. 2001. Mapping fire regimes across time and space: understanding coarse and fine-scale fire patterns. International Journal of Wildland Fire. 10(3-4): 329-342.

Murphy, K.A.; Reynolds, J.H.; Koltun, J.M. 2008. Evaluating the ability of the differenced normalized burn ratio (dNBR) to predict ecologically significant burn severity in Alaskan boreal forests. International Journal of Wildland Fire. 17(4): 490-499.

Murphy, P.J.; Mudd, J.P.; Stocks, B.J.; Kasischke, E.S.; Barry, D.; Alexander, M.E.; French, N.H.F. 2000. Historical fire records in the North American boreal forest. In: Kasischke, E.S.; Stocks, B.J., eds. Fire, climate change, and carbon cycling in the boreal forest. New York: Springer-Verlag: 274-288. Chapter 15.

Nanson, G.C.; Beach, H.F. 1977. Forest succession and sedimentation on a meandering-river floodplain, northeast British Columbia, Canada. Journal of Biogeography. 4(3): 229-251.

Nash, C.H.; Johnson, E.A. 1996. Synoptic climatology of lightning-caused forest fires in subalpine and boreal forests. Canadian Journal of Forest Research. 26(10): 1859-1874.

Natcher, D.C.; Calef, M.; Huntington, O.; Trainor, S.; Huntington, H.P.; DeWilde, L.; Rupp, S.; Chapin III, F.S. 2007. Factors contributing to the cultural and spatial variability of landscape burning by native peoples of interior Alaska. Ecology and Society. 12(1): 7. http://www.ecologyandsociety.org/vol12/iss1/art7/. (23 January 2011).

Neary, D.G.; Klopatek, C.C.; DeBano, L.F.; Ffolliott, P.F. 1999. Fire effects on belowground sustainability: a review and synthesis. Forest Ecology and Management. 122(1-2): 51-71.

Neinstaedt, H.; Zasada, J.C. 1990. *Picea glauca* (Moench) Voss, white spruce. In: Burns, R.M.; Honkala, B.H., eds. Silvics of North America, Volume 1. Conifers. Washington, D. C.: U.S. Department of Agriculture, Forest Service: 204-226.

Nelson, J.L.; Zavaleta, E.S.; Chapin III, F.S. 2008. Boreal fire effects on subsistence resources in Alaska and adjacent Canada. Ecosystems. 11(1): 156-171.

Niklasson, M.; Granstrom, A. 2000. Numbers and sizes of fires: long-term spatially explicit fire history in a Swedish boreal landscape. Ecology. 81(6): 1484-1499.

Norum, R.A. 1982. Predicting wildfire behavior in black spruce forests in Alaska. Res. Note PNW-401. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station. 10 p.

Nowacki, G.J.; Spencer, P.; Brock, T.; Fleming, M.; Jorgenson, T. 2001. Ecoregions of Alaska and neighboring territories [Environmental]. Miscellaneous Investigations Series 1 Map. [Place of publication unknown]: U.S. Department of the Interior, U.S. Geological Survey.

Papineau, J.M. 2001. Wintertime temperature anomalies in Alaska correlated with ENSO and PDO. International Journal of Climatology. 21(13): 1577-1592.

Paragi, T.F.; Haggstrom, D.A. 2007. Short-term responses of aspen to fire and mechanical treatments in interior Alaska. Northern Journal of Applied Forestry. 24(2): 153-157.

Parisien, M.A.; Peters, V.S.; Wang, Y.H.; Little, J.M.; Bosch, E.M.; Stocks, B.J. 2006. Spatial patterns of forest fires in Canada, 1980-1999. International Journal of Wildland Fire. 15(3): 361-374.

Parisien, M.A.; Sirois, L. 2003. Distribution and dynamics of tree species across a fire frequency gradient in the James Bay region of Quebec. Canadian Journal of Forest Research. 33(2): 243-256.

Payette, S. 1992. Fire as a controlling process in the North American boreal forest. In: Shugart, H.H.; Leemans, R.; Bonan, G.B., eds. A systems analysis of the global boreal forest. Cambridge: Cambridge University Press: 144-169. Chapter 5.

Payette, S.; Filion, L. 1985. White spruce expansion at the tree line and recent climatic change. Canadian Journal of Forest Research. 15(1): 241-251.

Payette, S.; Morneau, C. 1993. Holocene relict woodlands at the eastern Canadian treeline. Quaternary Research. 39(1): 84-89.

Payette, S.; Morneau, C.; Sirois, L.; Desponts, M. 1989. Recent fire history of the northern Québec biomes. Ecology. 70(3): 656-673.

Peters, V.S.; Macdonald, S.E.; Dale, M.R.T. 2006. Patterns of initial versus delayed regeneration of white spruce in boreal mixedwood succession. Canadian Journal of Forest Research. 36(6): 1597-1609.

Podur, J.J.; Martell, D.L. 2009. The influence of weather and fuel type on the fuel composition of the area burned by forest fires in Ontario, 1996-2006. Ecological Applications. 19(5): 1246-1252.

Pojar, J. 1996. Environment and biogeography of the western boreal forest. Forestry Chronicle. 72(1): 51-58.

Potter, C. 2004. Predicting climate change effects on vegetation, soil thermal dynamics, and carbon cycling in ecosystems of interior Alaska. Ecological Modelling. 175(1): 1-24.

Power, M.J.; Marlon, J.R.; Bartlein, P.J.; Harrison, S.P. 2010. Fire history and the global charcoal database: a new tool for hypothesis testing and data exploration. Palaeogeography, Palaeoclimatology, Palaeoecology. 291(1-2): 52-59.

Prentice, I.C. 1986. Vegetation responses to past climatic variation. Vegetatio. 67(2): 131-141.

Pyne, S.J.; Andrews, P.L.; Laven, R.D. 1996. Introduction to wildland fire. 2nd ed. New York: John Wiley & Sons, Inc. 769 p.

Quintillo, D.; Alexander, M.E.; Ponto, R.L. 1991. Spring fires in a semimature trembling aspen stand in central Alberta. Information Report NOR-X-323. Edmonton, AB: Environment Canada, Canadian Forestry Service, Northern Forestry Center. 30 p.

Quintillo, D.; Fahnestock, G.R.; Dubé, D.E. 1977. Fire behavior in upland jack pine: the Darwin Lake Project. Information Report NOR-X-174. Edmonton, AB: Environment Canada, Canadian Forestry Service, Northern Forestry Center. 49 p.

Quirk, W.A.; Sykes, D.J. 1971. White spruce stringers in a fire-patterned landscape in interior Alaska. In: Slaughter, C.W.; Barney, R.J.; Hansen, G.M., eds. Fire in the northern environment. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station: 179-197.

Racine, C.H.; Dennis, J.G.; Patterson, W.A. 1985. Tundra fire regimes in the Noatak River watershed, Alaska: 1956-83. Arctic. 38(3): 194-200.

Reap, R.M. 1991. Climatological characteristics and objective prediction of thunderstorms over Alaska. Weather and Forecasting. 6(3): 309-319.

Ritchie, J.C. 1982. The modern and late-Quaternary vegetation of the Doll Creek area, north Yukon, Canada. New Phytologist. 90(3): 563-603.

Ritchie, J.C. 1985. Late-Quaternary climatic and vegetational change in the lower Mackenzie basin, northwest Canada. Ecology. 66(2): 612-621.

Ritchie, J.C.; Hare, F.K. 1971. Late-Quaternary vegetation and climate near the Arctic tree line of northwestern North America. Quaternary Research. 1(3): 331-342.

Romme, W.H.; Turner, M.G.; Gardner, R.H.; Hargrove, W.W.; Tuskan, G.A.; Despain, D.G.; Renkin, R.A. 1997. A rare episode of sexual reproduction in Aspen (*Populus tremuloides* Michx) following the 1988 Yellowstone fires. Natural Areas Journal. 17(1): 17-25.

Rood, S.B.; Goater, L.A.; Mahoney, J.M.; Pearce, C.M.; Smith, D.G. 2007. Floods, fire, and ice: disturbance ecology of riparian cottonwoods. Canadian Journal of Botany. 85(11): 1019-1032.

Rothermel, R.C. 1972. A mathematical model for predicting fire spread in wildland fuels. Res. Pap. INT-115. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 40 p.

Rothermel, R.C. 1983. How to predict the spread and intensity of forest and range fires. Gen. Tech. Rep. INT-143. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 161 p.

Rowe, J.S.; Bergsteinsson, J. L.; Padbury, G.A.; Hermesh, R. 1974. Fire studies in the Mackenzie Valley. ALUR 73-74-61. Ottawa, ON: Department of Indian Affairs and Northern Development. 123 p.

Rowe, J.S.; Scotter, G.W. 1973. Fire in the boreal forest. Quaternary Research. 3: 444-464.

Rowe, J.S.; Spittlehouse, D.; Johnson, E.; Jasieniuk, M. 1975. Fire studies in the upper Mackenzie Valley and adjacent Precambrian uplands. ALUR 74-75-61. Ottawa, ON: Department of Indian Affairs and Northern Development. 128 p.

Rupp, T.S.; Starfield, A.M.; Chapin III, F.S. 2000a. A frame-based spatially explicit model of subarctic vegetation response to climatic change: comparison with a point model. Landscape Ecology. 15(4): 383-400.

Rupp, T.S.; Chapin III, F.S.; Starfield, A.M. 2000b. Response of subarctic vegetation to transient climatic change on the Seward Peninsula in north-west Alaska. Global Change Biology. 6(5): 541-555.

Rupp, T.S.; Chapin III, F.S.; Starfield, A.M. 2001. Modeling the influence of topographic barriers on treeline advance at the forest-tundra ecotone in northwestern Alaska. Climatic Change. 48(2-3): 399-416.

Rupp, T.S.; Chen, X.; Olson, M.; McGuire, A.D. 2007. Sensitivity of simulated boreal fire dynamics to uncertainties in climate drivers. Earth Interactions. 11: 1-21.

Rupp, T.S.; Starfield, A.M.; Chapin III, F.S.; Duffy, P. 2002. Modeling the impact of black spruce on the fire regime of Alaskan boreal forest. Climatic Change. 55: 213-233.

Ryan, K.C. 2002. Dynamic interactions between forest structure and fire behavior in boreal ecosystems. Silva Fennica. 36(1): 13-39.

Schimmel, J.; Granström, A. 1996. Fire severity and vegetation response in the boreal Swedish forest. Ecology. 77(5): 1436-1450.

Schimmel, J.; Granström, A. 1997. Fuel succession and fire behavior in the Swedish boreal forest. Canadian Journal of Forest Research. 27(8): 1207-1216.

Scott, P.A.; Hansell, R.I.C.; Fayle, D.C.F. 1987. Establishment of white spruce populations and responses to climatic change at the treeline, Churchill, Manitoba, Canada. Arctic and Alpine Research. 19(1): 45-51.

Senkowsky, S. 2001. A burning interest in boreal forests: researchers in Alaska link fires with climate change. Bioscience. 51(11): 916-921.

Serreze, M.C.; Box, J.E.; Barry, R.G.; Walsh, J.E. 1993. Characteristics of Arctic synoptic activity, 1952-1989. Meteorology and Atmospheric Physics. 51(3-4): 147-164.

Simard, M.; Lecomte, N.; Bergeron, Y.; Bernier, P.Y.; Paré, D. 2007. Forest productivity decline caused by successional paludification of boreal soils. Ecological Applications. 17(6): 1619-1637.

Sirois, L.; Payette, S. 1991. Reduced postfire tree regeneration along a boreal forest-forest-tundra transect in northern Quebec. Ecology. 72(2): 619-627.

Skinner, W.R.; Flannigan, M.D.; Stocks, B.J.; Martell, D.L.; Wotton, B.M.; Todd, J.B.; Mason, J.A.; Logan, K.A.; Bosch, E.M. 2002. A 500 hPa synoptic wildland fire climatology for large Canadian forest fires, 1959-1996. Theoretical and Applied Climatology. 71(3-4): 157-169.

Skinner, W.R.; Stocks, B.J.; Martell, D.L.; Bonsal, B.; Shabbar, A. 1999. The association between circulation anomalies in the mid-troposphere and area burned by wildland fire in Canada. Theoretical and Applied Climatology. 63(1-2): 89-105.

Slaughter, C.W.; Viereck, L.A. 1986. Climate characteristics of the taiga in interior Alaska. In: van Cleve, K.; Chapin III, F.S.; Flanagan, P.W.; Viereck, L.A.; Dyrness, C.T., eds. Forest ecosystems in the Alaskan taiga. New York: Springer-Verlag: 9-21. Chapter 2.

Smithwick, E.A.H.; Mack, M.C.; Turner, M.G.; Chapin III, F.S.; Zhu, J.; Balser, T.C. 2005. Spatial heterogeneity and soil nitrogen dynamics in a burned black spruce forest stand: distinct controls at different scales. Biogeochemistry. 76(3): 517-537.

Soja, A.J.; Tchebakova, N.M.; French, N.H.F.; Flannigan, M.D.; Shugart, H.H.; Stocks, B.J.; Sukhinin, A.I.; Parfenova, E.I.; Chapin III, F.S.; Stackhouse, P.W. 2007. Climate-induced boreal forest change: predictions versus current observations. Global and Planetary Change. 56(3-4): 274-296.

Soverel, N.O.; Perrakis, D.D.B.; Coops, N.C. 2010. Estimating burn severity from Landsat dNBR and RdNBR indices across western Canada. Remote Sensing of Environment. 114(9): 1896-1909.

Stocks, B.J. 1987. Fire behavior in immature jack pine. Canadian Journal of Forest Research. 17(1): 80-86.

Stocks, B.J. 1989. Fire behavior in mature jack pine. Canadian Journal of Forest Research. 19(6): 783-790.

Stocks, B.J.; Alexander, M.E.; Wotton, B.M.; Stefner, C.N.; Flannigan, M.D.; Taylor, S.W.; Lavoie, N.; Mason, J.A.; Hartley, G.R.; Maffey, M.E.; Dalrymple, G.N.; Blake, T.W.; Cruz, M.G.; Lanoville, R.A. 2004. Crown fire behaviour in a northern jack pine – black spruce forest. Canadian Journal of Forest Research. 34(8): 1548-1560.

Stocks, B.J.; Fosberg, M.A.; Lynham, T.J.; Mearns, L.; Wotton, B.M.; Yang, Q.; Jin, J.Z.; Lawrence, K.; Hartley, G.R.; Mason, J.A.; McKenney, D.W. 1998. Climate change and forest fire potential in Russian and Canadian boreal forests. Climatic Change. 38(1): 1-13.

Stocks, B.J.; Mason, J.A.; Todd, J.B.; Bosch, E.M.; Wotton, B.M.; Amiro, B.D.; Flannigan, M.D.; Hirsch, K.G.; Logan, K.A.; Martell, D.L.; Skinner, W.R. 2002. Large forest fires in Canada, 1959-1997. Journal of Geophysical Research. 108, 8149. DOI: 10.1029/2001JD000484.

Strong, W.L. 2009. *Populus tremuloides* Michx. postfire stand dynamics in the northern boreal-cordilleran ecoclimatic region of central Yukon Territory, Canada. Forest Ecology and Management. 258(7): 1110-1120.

Suffling, R.; Molin, J.D. 1982. Estimating past forest age distributions and disturbance rates in northwestern Ontario – a demographic approach. Journal of Environmental Management. 14(1): 45-56.

Swetnam, T.W.; Allen, C.D.; Betancourt, J.L. 1999. Applied historical ecology: using the past to manage for the future. Ecological Applications. 9(4): 1189-1206.

Tande, G.F. 1979. Fire history and vegetation pattern of coniferous forests in Jasper National Park, Alberta. Canadian Journal of Botany. 57(18): 1912-1931.

Taylor, A.H. 2000. Fire regimes and forest changes in mid and upper montane forests of the southern Cascades, Lassen Volcanic National Park, California, USA. Journal of Biogeography. 27(1): 87-104.

Timoney, K.P.; Wein, R.W. 1991. The areal pattern of burned tree vegetation in the subarctic region of northwestern Canada. Arctic. 44(3): 223-230.

Tinner, W.; Bigler, C.; Gedye, S.; Gregory-Eaves, I.; Jones, R.T.; Kaltenrieder, P.; Krähenbühl, U.; Hu, F.S. 2008. A 700-year paleoecological record of boreal ecosystem responses to climatic variation from Alaska. Ecology. 89(3): 729-743.

Tinner, W.; Hu, F.S.; Beer, R.; Kaltenrieder, P.; Scheurer, B.; Krähenbühl, U. 2006. Postglacial vegetational and fire history: pollen, plant macrofossil and charcoal records from two Alaskan lakes. Vegetation History and Archaeobotany. 15(4): 279-293.

Todd, S.K.; Jewkes, H.A. 2006. Wildland fire in Alaska: a history of organized fire suppression and management in the last frontier. Bulletin No. 114. Fairbanks, AK: University of Alaska Fairbanks, Agricultural and Forestry Experiment Station. 63 p.

Trigg, W.M. 1971. Fire season climatic zones of mainland Alaska. Res. Pap. PNW-126. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station. 12 p.

Turetsky, M.R.; Mack, M.C.; Hollingsworth, T.N.; Harden, J.W. 2010. The role of mosses in ecosystem succession and function in Alaska's boreal forest. Canadian Journal of Forest Research. 40(7): 1237-1264.

Turner, M.G.; Romme, W.H.; Gardner, R.H.; Hargrove, W.W. 1997. Effects of fire size and pattern on early succession in Yellowstone National Park. Ecological Monographs. 67(4): 411-433.

Tymstra, C.; Flannigan, M.D.; Armitage, O.B.; Logan, K. 2007. Impact of climate change on area burned in Alberta's boreal forest. International Journal of Wildland Fire. 16(2): 153-160.

van Cleve, K.; Chapin III, F.S.; Dyrness, C.T.; Viereck, L.A. 1991. Element cycling in taiga forests: state-factor control. Bioscience. 41(2): 78-88.

van Cleve, K.; Chapin III, F.S.; Flanagan, P.W.; Viereck, L.A.; Dyrness, C.T. 1986. Forest ecosystems in the Alaskan taiga: a synthesis of structure and function. New York: Springer-Verlag.

van Cleve, K.; Dyrness, C.T.; Viereck, L.A.; Fox, J.; Chapin III, F.S.; Oechel, W. 1983. Taiga ecosystems in interior Alaska. Bioscience. 33(1): 39-44.

van Cleve, K.; Viereck, L.A. 1981. Forest succession in relation to nutrient cycling in the boreal forest of Alaska. In: West, D.C.; Botkin, D.B.; Shugart, H.H., eds. Forest succession: concepts and application. New York: Springer-Verlag: 185-211.

Van Wagner, C.E. 1983. Fire behavior in northern conifer forests and shrublands. In: Wein, R.W.; MacLean, D.A., eds. The role of fire in northern circumpolar ecosystems. New York: John Wiley & Sons Ltd.: 65-80. Chapter 4.

Van Wagner, C.E. 1988. The historical pattern of annual burned area in Canada. Forestry Chronicle. 64(3): 182-185.

Verbyla, D.; Lord, R. 2008. Estimating post-fire organic soil depth in the Alaskan boreal forest using the normalized burn ratio. International Journal of Remote Sensing. 29(13): 3845-3853.

Viereck, L.A. 1970. Forest succession and soil development adjacent to the Chena River in interior Alaska. Arctic and Alpine Research. 2: 1-26.

Viereck, L.A. 1973. Wildfire in the taiga of Alaska. Quaternary Research. 3: 465-495.

Viereck, L.A. 1983. The effects of fire in black spruce ecosystems of Alaska and northern Canada. In: Wein, R.W.; MacLean, D.A., eds. The role of fire in northern circumpolar ecosystems. New York: John Wiley & Sons Ltd.: 201-220. Chapter 11.

Viereck, L.A.; Dyrness, C.T. 1979. Ecological effects of the Wickersham Dome fire near Fairbanks, Alaska. Gen. Tech. Rep. PNW-90. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station. 71 p.

Viereck, L.A.; Dyrness, C.T.; Batten, A.R.; Wenzlick, K.J. 1992. The Alaska vegetation classification. Gen. Tech. Rep. PNW-286. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 278 p.

Viereck, L.A.; Dyrness, C.T.; Foote, M.J. 1993. An overview of the vegetation and soils of the floodplain ecosystems of the Tanana River, interior Alaska. Canadian Journal of Forest Research. 23(5): 889-898.

Viereck, L.A.; Dyrness, C.T.; van Cleve, K.; Foote, M.J. 1983. Vegetation, soils, and forest productivity in selected forest types in interior Alaska. Canadian Journal of Forest Research. 13(5): 703-720.

Viereck, L.A.; Schandelmeier, L.A. 1980. Effects of fire in Alaska and adjacent Canada – a literature review. Technical Report 6. Anchorage, AK: U.S. Department of the Interior, Bureau of Land Management, Alaska State Office. 124 p.

Viereck, L.A.; van Cleve, K.; Dyrness, C.T. 1986. Forest ecosystem distribution in the taiga environment. In: van Cleve, K.; Chapin III, F.S.; Flanagan, P.W.; Viereck, L.A.; Dyrness, C.T., eds. Forest ecosystems in the Alaskan taiga. New York: Springer-Verlag: 22-43. Chapter 3.

Wahrhaftig, C. 1965. Physiographic divisions of Alaska. Professional Paper 482. [Place of publication unknown]: U.S. Department of the Interior, U.S. Geological Survey. 52 p.

Walker, L.R.; Zasada, J.C.; Chapin III, F.S. 1986. The role of life history processes in primary succession on an Alaskan floodplain. Ecology. 67(5): 1243-1253.

Wang, G.G. 2002. Fire severity in relation to canopy composition within burned boreal mixedwood stands. Forest Ecology and Management. 163(1-3): 85-92.

Wang, G.G. 2003. Early regeneration and growth dynamics of *Populus tremuloides* suckers in relation to fire severity. Canadian Journal of Forest Research. 33(10): 1998-2006.

Wang, X.L.; Cumming, S.G. 2010. Configuration dynamics of boreal forest landscapes under recent fire and harvesting regimes in western Canada. Landscape Ecology. 25(9): 1419-1432.

Ward, P.C.; Mawdsley, W. 2000. Fire management in the boreal forest of Canada. In: Kasischke, E.S.; Stocks, B.J., eds. Fire, climate change, and carbon cycling in the boreal forest. New York: Springer-Verlag: 66-84. Chapter 5.

Ward, P.C.; Tithecott, A.G.; Wotton, B.M. 2001. Reply – A re-examination of the effects of fire suppression in the boreal forest. Canadian Journal of Forest Research. 31(8): 1467-1480.

Wein, R.W. 1976. Frequency and characteristics of Arctic tundra fires. Arctic. 29(4): 213-222.

Wein, R.W.; MacLean, D.A. 1983. The role of fire in northern circumpolar ecosystems. New York: John Wiley & Sons, Inc. 322 p.

Wein, R.W.; Moore, J.M. 1977. Fire history and rotations in the New Brunswick Acadian forest. Canadian Journal of Forest Research. 7: 285-294.

Wein, R.W.; Moore, J.M. 1979. Fire history and recent fire rotation periods in the Nova Scotia Acadian forest. Canadian Journal of Forest Research. 9(2): 166-178.

Weir, J.M.H.; Johnson, E.A. 1998. Effects of escaped settlement fires and logging on forest composition in the mixedwood boreal forest. Canadian Journal of Forest Research. 28(3): 459-467.

Weir, J.M.H.; Johnson, E.A.; Miyanishi, K. 2000. Fire frequency and the spatial age mosaic of the mixed-wood boreal forest in western Canada. Ecological Applications. 10(4): 1162-1177.

Wirth, C.; Lichstein, J.W.; Dushoff, J.; Chen, A.; Chapin III, F.S. 2008. White spruce meets black spruce: dispersal, postfire establishment, and growth in a warming climate. Ecological Monographs. 78(4): 489-505.

Wotton, B.M.; Flannigan, M.D. 1993. Length of the fire season in a changing climate. Forestry Chronicle. 69(2): 187-192.

Wotton, B.M.; Nock, C.A.; Flannigan, M.D. 2010. Forest fire occurrence and climate change in Canada. International Journal of Wildland Fire. 19(3): 253-271.

Wright, H.E.; Kutzbach, J.; Webb, T.; Ruddiman, W.F.; Street-Perrott, F.A.; Bartlein, P. 1993. Global climate since the last glacial maximum. Minneapolis, MN: University of Minnesota Press. 569 p.

Wulder, M.A.; White, J.C.; Alvarez, F.; Han, T.; Rogan, J.; Hawkes, B. 2009. Characterizing boreal forest wildfire with multi-temporal Landsat and LIDAR data. Remote Sensing of Environment. 113(7): 1540-1555.

Xiao, J.F.; Zhuang, Q.L. 2007. Drought effects on large fire activity in Canadian and Alaskan forests. Environmental Research Letters. 2(4): doi:10.1088/1748-9326/2/4/044003.

Yalcin, K.; Wake, C.R.; Kreutz, K.J.; Whitlow, S.I. 2006. A 1000-yr record of forest fire activity from Eclipse Icefield, Yukon, Canada. Holocene. 16(2): 200-209.

Yarie, J. 1981. Forest fire cycles and life tables: a case study from interior Alaska. Canadian Journal of Forest Research. 11(3): 554-562.

Yarie, J.; Viereck, L.; van Cleve, K.; Adams, P. 1998. Flooding and ecosystem dynamics along the Tanana River. Bioscience. 48(9): 690-695.

Youngblood, A.; Max, T.A. 1992. Dispersal of white spruce seed on Willow Island in interior Alaska. Res. Pap. RP-PNW-443. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 17 p.

Zackrisson, O. 1977. Influence of forest fires on north Swedish boreal forest. Oikos. 29(1): 22-32.

Zasada, J.C.; Gregory, R.A. 1969. Regeneration of white spruce with reference to interior Alaska; a literature review. Res. Pap. PNW-79. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station. 37 p.

Zasada, J.C.; Norum, R.A.; Teutsch, C.E.; Densmore, R. 1987. Survival and growth of planted black spruce, alder, aspen and willow after fire on black spruce feather moss sites in interior Alaska. Forestry Chronicle. 63(2): 84-88.

Zasada, J.C.; Norum, R.A.; Van Veldhuizen, R.M.; Teutsch, C.E. 1983. Artificial regeneration of trees and tall shrubs in experimentally burned upland black spruce/feather moss stands in Alaska. Canadian Journal of Forest Research. 13(5): 903-913.

Zasada, J.C.; Phipps, H.M. 1990. *Populus balsamifera* L., Balsam Poplar. In: Burns, R.M.; Honkala, B.H., eds. Silvics of North America, Volume 2. Hardwoods. Washington, D.C.: U.S. Department of Agriculture, Forest Service: 518-529.

Zhang, Q.; Chen, W. 2007. Fire cycle of the Canada's boreal region and its potential response to global change. Journal of Forestry Research. 18(1): 55-61.

List of figures and tables

Appendix 1: Alaskan Fire History Study Summaries

Reference: Barney, R.J. 1971. Wildfires in Alaska - some historical and projected effects and aspects. In: Slaughter, C.W.; Barney, R.J.; Hansen, G.M., eds. Fire in the northern environment. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station: 51-59.

Introduction

The purpose of this paper is to present quantitative measures of wildfires in Alaska in order to draw conclusions about their causes and extent. Fire statistics for the period 1940-69 were analyzed to provide details of fire occurrence during the period of fire suppression. Anecdotal data and historical accounts were used in conjunction with fire statistics to speculate how fire regimes in Alaska appeared during pre-settlement and pre-suppression time periods. Additionally, this paper discusses the lack of information concerning fire regimes in Alaska and the detrimental effect this has on fire management policy.

Methods

To assess the scope of wildfires in Alaska this paper reviewed previously compiled fire statistics (Barney 1969, 1971, Hardy 1963, Lutz 1956) and discussed them in the context of historical accounts of wildfires.

Results

Both native Alaskans and western settlers were acknowledged as possible significant sources of wildfire as both cultures used fire for insect control, land clearing, hunting, and signaling. It was likely that the number of human-caused ignitions increased during the settlement period which began in the late 18th century after gold was discovered in Alaska. During early settlement, the population of Alaska increased dramatically, thus increasing the number of ignitions, without any sort of organized fire suppression. Early fire records for the state cover the period 1893-1937. These records are informal and include only 19 fires that burned a total of 2.5 million ha. Other estimates put the annual acreage burn during this period at 1 million acres. Early formal fire statistics for the period 1940-49 likely underestimated acreage burned because fire detection was limited mainly to roads which only covered a tiny fraction of the state. Nevertheless these records indicated that the annual acreage burned during this time was approximately 485 000 ha. Since these estimates are all likely to be conservative due to the lack of comprehensive detection, annual acreage burned of 0.6-1.0 million ha was more likely.

The average annual acreage burned during the period of formal fire records (1940-69) was 405 000 ha. This was likely to be a better estimate because detection methods were more comprehensive. This data also indicates that lightning-caused fires were responsible for only 30 percent of all fire starts but 78 percent of the area burned. Fire records also show two general trends during the past 30 years: the number of fires has increased but the annual area burned has decreased. A likely reason for these trends was increasing numbers of human-caused fires as the population grew, and decreasing size of the fires as suppression activities became more effective. These statistics also show that nearly all these fires were confined to interior Alaska.

Based on the rough calculation that 40.5 million ha of land has burned during the past 70 years, the author believes that much of interior Alaska would have burned over during the past 200-250 years. This estimate appeared low relative to reported rotation ages for typical forest types: 100-150 years for white spruce, 80-100 years for paper birch, and 60-80 years for aspen.

Two primary detrimental impacts of wildfire on natural resources were proposed. One was the negative impact of fires on winter range of caribou and the other was the possible interference fires have on succession on productive upland sites from relatively low value hardwoods to high value white spruce. On the other hand two possible positive impacts of fire include the creation of moose range and the preventing paludification of riparian stands of white spruce.

Discussion & Conclusions

The author concluded by stating that a more accurate quantification of the boreal fire regime is required for the successful management of the ecosystem. This paper summarizes the current state of knowledge regarding fire regimes in Alaska, in particular for the boreal forest.

Reference: Christiansen, J.S. 1988. A spruce-lichen woodland in northern Alaska: post-fire regeneration and community dynamics. Seattle, WA: University of Washington. 94 p. M.S. thesis.

Introduction

This research assessed community dynamics and age structure of a spruce-lichen woodland in Alaska. Though common in Canada, spruce-lichen woodlands occur infrequently in Alaska. The communities exist on sandy glacial deposits at the northern limit of tree growth. These two conditions only occur along the southern edge of the Brooks Range. The objectives of this research included describing the vegetation of a spruce-lichen woodland, documenting fire history, and estimating post-fire regeneration.

Methods

The study site was located on a terminal moraine at the south end of Walker Lake, on the south slopes of the central Brooks Range, Alaska. The study area is at the northern limit of tree growth and spruce-lichen woodland tree species composition was a gradient of combinations of black and white spruce that included pure stands of both species. The shrub layer was sparse and the forest floor cover was primarily fruticose lichens. Other vegetation types within the study area included black spruce-feathermoss forest, mixed boreal forest, aspen groves, and shrub tundra dominated by shrub birch.

Fire history data was collected by opportunistic sampling of fire-scarred trees to determine fire dates and the oldest live trees to determine stand initiation dates. Seven vegetation plots were also established to assess vegetation composition, tree age distributions, and soil characteristics. Increment cores or basal cross sections were collected from all trees in each plot. Samples were not cross-dated to account for missing growth rings.

Results

Two fires in the study area were identified. A fire east of the Walker Lake outlet river burned in 1913 and a fire west of the Walker Lake outlet river burned in 1891. The lack of double fire scares on trees along the shared boundary of the two fires suggested the two fires did not overlap. Trees that survived the 1891 and 1913 fires (legacy trees) established between 1840 and 1870 with many establishing in the 1850's. The basal diameter of the fire survivors at the time of the 1891 and 1913 fires was small and indicated they originated from a disturbance in the 1840's.

Vegetation and soil characteristics were extensively detailed for each plot and reflected the mosaic of vegetation composition and structure typically found at the northern limit of tree growth in diverse topography.

Tree regeneration following fire in plots dominated by black spruce occurred primarily within the first 35-50 years after fire and peaked 10-20 years following fire. At the time of sampling, tree establishment had ended in most plots and regeneration through layering was uncommon. At plots dominated by white spruce, tree establishment following was longer, approximately 50-80 years and the regeneration period lacked the strong pulse of establishment observed in black spruce plots. Tree establishment was continuing at the time of sampling for half of the plots and regeneration of the black spruce component through layering was occurring at one plot.

Discussion & Conclusions

Variability of legacy tree frequency and organic layer depth suggested the area was impacted by a wide range of fire severity. The frequency of trees that survived the 1981 fire was larger relative to the 1913 fire, indicating severity for the latter was higher. Possible reasons burn severity was higher for the 1913 fire included drier weather or increased stand flammability. The pre-fire stand would have been approximately 70 years in 1913 vs. 50 years in 1891. Previous research (Rowe et al. 1975) has suggested that a 20 year age difference could influence flammability, the increase in flammability may have resulted in higher fire severity.

Establishment of black spruce was more rapid than white spruce. Black spruce has several adaptations to fire that would allow this species to reestablish faster than white spruce. First, black spruce has semi-serotinous cones that store seeds for several years and can continue to release seed for up to eight years following fire (Black and Bliss 1980). This provides a much larger seed source relative to white spruce which has no mechanism for storing seeds. Second fire-killed black spruce release seed consistently throughout the burned area while white spruce seed sources must travel from live trees located along the burn edge or from isolated legacy trees (Viereck 1973). Because spruce seeds are heavy they do not travel far from parent trees. Stand age distribution varied widely among plots and was attributed to factors related to burn severity including: frequency of surviving trees, soil moisture holding capacity and depth of the organic mat. There were two plots where burn severity was low, one was dominated by black spruce and one was dominated by white spruce. In the low severity black spruce plot, the pulse of establishment was weak and the establishment period was longer. For the low severity white spruce plot, there was a higher density of regenerating trees, growth was slower and the establishment period was shorter.

Wide fluctuations of moisture and temperature for charred organic (Vincent 1965) matter and dry conditions of exposed mineral soil (Smith 1951, Vincent 1965) combined with slow radical growth for black spruce (Foster 1985) likely has an adverse influence on seedling establishment immediately following fire. Initial colonization of burned sites by pioneer mosses and lichen (Kershaw and Rouse 1971) may ameliorate site conditions for regenerating seedlings (Smith 1951, Vincent 1965) and could explain delayed peaks in establishment observed. Potential reasons for low or no levels of tree establishment in mature stands include lack of sunlight (Goldstein 1981), root competition (Cowles 1982), and reductions in seedbed quality due to the lichen mat (Brown and Mikola 1974, Cowles 1982).

Research conducted at higher elevations surrounding Walker Lake indicated levels of recruitment in stands of white spruce of undetermined origin remained relatively high at the time of sampling (Goldstein 1981). The higher levels of continuing recruitment reported by Goldstein (1981) was possibly due to more favorable climatic conditions at high elevations.

Reference: De Volder, A. 1999. Fire and climate history of lowland black spruce forests, Kenai National Wildlife Refuge, Alaska. Flagstaff, AZ: Northern Arizona University. 128 p. M.S. thesis.

Introduction

A time-since-last-fire (TSLF) map was created to determine fire history for the Kenai National Wildlife Refuge (NWR), Alaska for the period 1700-1996. This research provides the first fine-scale spatial account of fire history for boreal forests in Alaska. The fire regime was quantitatively assessed and detailed descriptions are provided for each fire.

Methods

Samples were collected at 171 sites located subjectively to maximize the likelihood of finding fire-scarred trees. The study area included 98 200 ha of lowland black spruce forest within the Kenai NWR. Two measures of fire frequency (*mFRI* and fire cycle) were calculated for stratified physiographic regions (mountainous, rain shadow, and lakes regions. Fire frequency calculations excluded data from the second half of the 20th century because a fire in 1947 burned greater than 80 percent of the study area and thus violated an assumption of the fire cycle calculation that states individual fires cannot exceed 1/3 of the study area. The TSLF map, which represents stand ages in 1946 was produced by reconciling three types of dendrochronological data (fire scar dates, outer ring dates from fire-killed trees, and age distributions) with forest stand boundaries interpreted from aerial photos pre-dating the 1947 fire. Dendrochronological data was verified by visually cross-dating each sample with a master chronology and assessing accuracy with the COFECHA program. The master chronology developed for this study site spans 300 years for lowland black spruce. Detailed fire records maintained since 1940 provided an accurate record of fire history after the 1947 fire and were used to describe fires during this period. The 1946 TSLF map extends fire history back to 1700.

Results

A total of 12 pre-1947 fires were dated, although evidence suggests additional fires burned through the study area. Data was not sufficient to confirm dates and perimeters for these fires, thus the fire history described here is conservative. Fire records indicated that 6 fires occurred after 1940. Fire dates, acreage burned, and percentage of study area burned are listed on table 3. For the period 1700-1996, slightly more than 50 percent of fires occurred early in the growing season (May and June) and the rest occurred in the middle to late portion of the growing season (July through early September). The pre-1947 *mFRI* for the entire study area was 89 years. For the three physiographic regions, mountainous, rain shadow and lakes, the *mFRI* was 132, 88, and 69 years, respectively (table 3). Plotting cumulative area burned based on the 1946 TSLF map produced a curve that closely approximated the Weibull distribution. Living cohorts in 1946 represented fire dates from 1768 to 1898 and the fire cycle was 42 years.

A review of each pre-20th century fire indicates that estimates of acreage burned are conservative. Two factors increase the likelihood of underestimating total area burned: low-intensity fire and loss of evidence over time. As fire intensity in boreal forest decreases the regenerating age cohort becomes less robust or may be absent. Thus, low-intensity fires are less likely to be identified. However, the affect of this factor is likely to be small. Only the 1849 fire appeared to burn at low intensities as revealed by multi-aged cohorts in the burn area. Data indicated all other fires were stand replacing events. Loss of fire evidence over time is a potentially greater source of error for this study. Of the 12 pre-20th century fires recorded, burned area estimates of 11 were considered minimum and, except in the case of the 1849 fire, degradation of dendrochronological fire records over time was the only reason for potential underestimates.

Discussion & Conclusions

Statistical tests of significance indicated there was no difference in *mFRI* between the lakes region and the rain shadow region. The mountainous region could not be statistically compared to other regions because the number of intervals was too low. The data collected from the mountainous region suggested that fire frequency is much lower than adjacent lowlands to the west (lakes and rain shadow regions).

Temporal distribution of historic fire frequency indicates two periods of relatively high fire frequency (1700-62 and 1828-98) and two periods of relatively low fire frequency (1763-1827 and 1899-1946). Explanations for temporal variations in fire frequency include sampling error, climate change, and anthropogenic activities. The early to mid 18th century corresponds to a period of higher temperatures during the Little Ice Age (LIA) which may explain the higher fire frequency observed. For the period 1763-1827, only one small fire was recorded. Sampling bias due lack of fire evidence was an unlikely explanation for low fire frequency during this period because fire frequency was higher for earlier dates. Climate change is a possible explanation as the late 18th and early 19th century was the coldest period of the LIA and climate was likely wetter as well. For the period 1828-98 nine fires were recorded. This period coincides with early settlement and moderating temperatures, but it is not possible to separate the relative influences of these two factors. There was an absence of fires from 1900 through 1946; this was likely due to sampling bias. Trees regenerating from fires that burned between 1900 and 1946 would have been seedlings or saplings at the time of the 1947 fire and would have fallen and decomposed quickly due to the shallow root systems and small diameter stems, respectively thus leaving little evidence of their presence during sample collection.

Superposed epoch analysis (SEA) was used to examine potential relationships between fire events and temperature trends (Briffa et al. 1992). SEA yielded no significant relationship between fire frequency and temperature or ring width indices. A lack of correlation between temperature and fire frequency is not uncommon in boreal forests; weather parameters that generally have the highest impact on fire behavior are precipitation and relative humidity.

The fire regime described by this research is typical for western boreal black spruce forest. The author cautions interpretation of reported fire regime data since not all fires during the study period were included in the analysis and early settlement may have altered fire regimes. The increasing temperatures on the Kenai Peninsula and growing source of human ignitions will likely have an impact on the future fire regime on the Kenai Peninsula. Understanding the human and climate related impacts on fire regimes are important to future management of disturbances on the Kenai NWR.

Reference: Earle, C.J.; Brubaker, L.B.; Anderson, P.M. 1996. Charcoal in northcentral Alaskan lake sediments: relationships to fire and late-Quaternary vegetation history. Review of Palaeobotany and Palynology. 92(1-2): 83-95.

Introduction

This study had two objectives: identify the relationship between lake sediment charcoal and local fires, and assess patterns of fire, vegetation, and climate for the past 14 000 years. Reported dates are uncalibrated.

Methods

For each objective there was a separate study area. The study area for the first objective, referred to as the modern study area, was located in the Upper Kuskokwim River drainage and included 29 lakes. The study area for the second objective was located at Sithylemenkat Lake, located in the Kanuti Flats. Sithylemenkat Lake is up to 12 m deep and is 9 km^2 in area. Forest types surrounding Sithylemenkat Lake included black spruce muskeg on poorly drained sites and white spruce, paper birch, and quaking aspen on well drained sites.

To assess the relationship between charcoal in lake sediment and local fires, shallow (less than 2 cm) sediment samples were collected from the sediment-water interface at each lake in the modern study area. The drainage surrounding each lake was classified into three burn categories (unburned, partially burned, and extensively burned) based on the percentage of area burned (0, less than 50, and greater than 50 percent) in recent fires (1986 and 1990). Charcoal particles within each sample were quantified by image analysis software calibrated to identify and calculate charcoal frequency and area for six size classes. To assess historic vegetation and fire activity at Sithylemenkat Lake; fossil pollen and charcoal were quantified. Samples were collected from the sediment core at 6 cm intervals and each sample represented 120-260 years of sediment accumulation.

Results

In the modern study area, particle frequency for the three smallest charcoal size classes was dramatically higher relative to the three largest size classes; however, charcoal area for all size classes was roughly the same. On average, charcoal characteristics (i.e. frequency and area) for the three smallest size classes increased from unburned to extensively burned categories. In the three largest size classes; median charcoal characteristics were lowest for the partially burned category, followed by unburned and extensively burned categories. Most importantly, for all size classes, there was considerable overlap in the range charcoal characteristics among the three burn categories. Information regarding the best use of charcoal particle data to infer fire activity was extended to the analysis of fossil charcoal from Sithylemenkat Lake.

For the Sithylemenkat Lake, fire activity appeared to respond to changes in climate and vegetation. From 14 000-10 000 BP, there were strong charcoal peaks, suggesting fire was frequent. Based on fossil pollen data, vegetation during this period was dominated by shrub birch tundra. From 10 000-9 500 BP, fossil pollen data indicated an expansion of balsam poplar and white spruce forest. This period of forest expansion coincided with a period of warmer than present temperatures (Anderson and Brubaker 1994). During this period charcoal peaks were still relatively frequent, indicating fire was still an important disturbance. Approximately 8000 BP white spruce pollen and charcoal frequency declined and alder pollen increased. These changes to the fire regime and vegetation were possibly influenced by increased effective moisture, caused by decreasing temperatures and/or increasing precipitation (Hu et al. 1993, Anderson and Brubaker 1994). Approximately 6500 BP fossil pollen indicated black spruce expanded. This vegetation shift was possibly influenced by climatic cooling (Anderson and Brubaker 1994). During this period there was a slight increase in charcoal frequency. Given it was unlikely that cooler climate increased fire frequency, the shift in fire frequency was attributed to high flammability black spruce (Viereck 1973).

Discussion & Conclusions

Results from the modern study area suggested a poor relationship between local fires and charcoal characteristics in lake sediments. While median trends were generally as expected, the amount of overlap among area burned categories revealed large amounts of variability within each burn category. A number of factors could explain the variation; the authors conclude that variability in fire behavior and distances between burned and unburned watersheds likely had the largest impact on observed variation among sites.

The results from Sithylemenkat Lake suggested fire regimes were influenced by both climate and vegetation. Both birch shrub tundra and black spruce forest are flammable fuel types and both supported high fire frequency. Drier and warmer weather for the period 10 000-9 500 BP likely influenced the fire regime both indirectly, through supporting a fuel type that would burn (white spruce and balsam poplar forest) and directly through weather that promoted dry fuels. The higher effective moisture from 8000-6500 BP coincided with decreased fire activity and possibly had a direct influence on the fire regime by increasing fuel moisture.

The results for charcoal characteristics conflict with results of similar research in the region (Hu et al. 1993). Though the fossil pollen history was relatively similar, Hu et al. (1993) found that charcoal content was greatest after 9000 BP while this research noted charcoal content was greatest prior to 9000 BP. This inconsistency may have been due to differences in charcoal particle size selected for analysis. Hu et al. (1993) analyzed larger charcoal particle sizes. It is feasible that conifers release larger charcoal particles during fires than herbs and deciduous shrubs. If this is true, the smaller particle sizes analyzed in this study would be biased towards fires occurring greater than 9000 BP when tundra was dominant and Hu et al.'s (1993) research would have been biased towards fires occurring less than 9000 BP when coniferous forest was the dominant vegetation type, thus explaining the difference in charcoal content between the two studies.

Reference: Fastie, C.L.; Lloyd, A.H.; Doak, P. 2002. Fire history and postfire forest development in an upland watershed of interior Alaska. Journal of Geophysical Research. 107: 8150, doi: 10. 1029/2001JD000570, [printed 108(D1), 2003].

Introduction

Fire history research was conducted in the Caribou-Poker Creek Research Watershed (CPCRW). There were two objectives for this research: to understand fire history and describe how forest composition has changed over the past 150 years. This is the first time spatially explicit fire history study in interior Alaska.

Methods

The study area is located approximately 50 miles north of Fairbanks, Alaska, and occupies three sub-basins within the CPCRW. Four types of data were collected: dated fire scars recovered from living trees, age distributions of live trees and fire-killed trees, outer ring dates from fire-killed trees, and dates of abrupt increases in radial growth. Multiple sampling schemes were used at 43 sites located across four sampling units: C4 (26 sites), P6 (13 sites), and C3 (three sites) sub-basins, and confluence of Caribou and Poker creeks (one site). Fire perimeters were mapped by overlaying dendrochronological data on aerial photographs (*c.* 1951), then mapping perimeters based on visible burn areas. Forests in the study area were dominated by black spruce forests on upper and north facing slopes, stands of paper birch with minor components of trembling aspen and black spruce on south facing slopes, and white spruce atop sparsely vegetated ridges and along drainages.

Results

Fire scar data from the C4 sub-basin recorded fires in 1896, 1909, and 1924 that burned 11.7, 3.7, and 78 percent of the sub-basin, respectively. The 1896 and 1909 fires were generally restricted to ridgetops. Age distribution data from living trees suggested stand recruitment continued for 2-4 decades following fire with the exception of south slopes burned in 1924 where the stand recruitment period was 6-7 decades. Establishment dates from fire-killed trees in the C4 sub-basin suggested they established following a disturbance prior to 1750.

Fire scar data from the P6 sub-basin recorded fires in 1902, 1924, 1925, and 1935. Living stands were only detected for the 1902 and 1924 fires and indicated the fires burned 29.1 and 17.5 percent of the sub-basin, respectively. These two fires burned primarily on ridgetops and age distributions of living trees suggested the post-fire recruitment period was 1-4 decades. Fire-killed trees regenerated following two disturbances in approximately 1790 and 1805. 53 percent of the P6 sub-basin was unburned during the 20th century; these included stands of black spruce on north facing sloped and muskeg that established between 1780 and 1800 and mixed spruce forest on south facing slopes that established between 1795 and 1810.

Data collected from the P3 sub-basin indicated two fires in 1905 and 1911. Recruitment dates from fire-scarred trees and other individuals that survived 20th century fires suggest the area had not been impacted by fire since at least 1750. Samples from the confluence of Caribou and Poker creeks indicated that a fire occurred in 1901.

Discussion & Conclusions

Due to intensive mining that occurred near the study area and fire suppression policy, 20th century fires likely do not reflect reference conditions. The first two gold prospectors arrived in the Fairbanks area in 1901 and the 1909, 1924, and 1925 fires may have been caused by mining activities. Fires documented by fire scar data that occurred prior to 1905 were combined with stand initiation data to estimate a reference condition *FRI* for the C4 and P6 sub-basins of 146 years and 102 years, respectively. Both these areas were located in black spruce forest.

Sampling techniques for this study were not able to capture low-intensity fires which may not be recorded in the dendrochronological record. The paper notes that two fires in the P6 basin (1924 and 1935) are not associated with any evidence of recruitment and may have burned at low intensities.

Two distinct fire regimes may exist within the study area: ridgetops and stringer forests. Three (1896, 1902, and 1909) of the five fires recorded in the C4 and P6 sub-basins were restricted to ridgetops. Ridgetops may be subject to higher fire frequency than other topographical locations because they are more prone to lightning, winds are stronger, and drainage is better. Age data from stringer forests suggested fire bypassed these stands. Stringer forests in the C4 sub-basin have been fire free for at least 250 years even though adjacent stands have burned more frequently.

Post-fire stands for multiple forest types closely resembled pre-fire stands suggesting self replacement is common. Self replacement occurred in black spruce and paper birch. The persistence of pure stands of paper birch may be attributable to frequent, low-intensity surface fires that suppress spruce establishment. Evidence for this fire regime is given by fire scars in paper birch stands. Though black spruce is a co-dominant in some stands of birch there is no indication that relay succession from paper birch to black spruce occurs. Stand data from the P6 sub-basin suggested a 250 year old paper birch stand with black spruce as a co-dominant has maintained its present composition for the past 75 years. Self-replacing paper birch and static composition in mixed-wood stands suggests hardwood to conifer relay succession (Dyrness et al. 1986, Van cleve et al. 1991) does not occur.

Reference: Gabriel, H.W.; Tande, G.F. 1983. A regional approach to fire history in Alaska. Technical Report 9. Anchorage, AK: U.S. Department of the Interior, Bureau of Land Management, Alaska State Office. 34 p.

Introduction

This research marked the first time fire statistics were used to calculate quantitative measures of fire frequency in Alaska. Individual fire records were mapped in relation to administrative, ecological, and meteorological boundaries. The results suggested Alaska contained several different fire regimes, however the data period was short (1957-79) and accuracy of fire records was inconsistent. The authors note that the fire regimes during the period analyzed had been influenced by fire suppression and settlement and it was not possible to assess the relative contributions of these activities on the fire regime.

Methods

The study area included all of Alaska except the southeast panhandle and the Aleutian Peninsula (figure 8). Fire records extended to 1940 but due to concerns about accuracy for earlier records, only the period 1957-79 was selected for analysis. Though errors were acknowledged for the fire records analyzed, this dataset was deemed sufficient for the purposes of the research. Analysis of the data included calculation of four quantitative measures: fires/million ha/23 years, percent area burned/23 years, mean fire size, and fire cycle. These measures were calculated for the entire study area and for geographic units including Alaska planning regions, physiographic provinces, physiographic sections (Wahrhaftig 1965), and fire weather zones. Spatial analysis of fire statistics included only fires started by lightning for the period 1957-79.

Results

Fire location maps depicted a stark difference in distribution of lightning-caused fires, which are concentrated in interior Alaska, and human-caused fires which are largely confined to major road corridors and population centers. Number of fires/23 years for each of the six Alaska planning regions revealed geographic differences in fire regimes. Planning regions centered in the center of the state had the highest number of fires/23 years while planning regions adjacent to the southern and northern coasts had the lowest. However these planning regions were large and did not follow natural boundaries of terrain and vegetation. Thus no further analysis was conducted using these geographic units.

Of the 12 physiographic provinces, 11 fell within the study area. Fire statistics calculated for each province indicated that number of fires/23 years was highest in interior Alaska and progressively decreased to the north and south of the Brooks and Alaska Ranges, respectively. Mean fire size does not follow the same spatial pattern as number of fires/23 years. Some coastal provinces have the highest mean fire size. The lack of correspondence of mean fire size with number of fires/23 years was attributed to the large variation in fire size. The number of fires recorded in the coastal provinces was low, thus just one or two very large fires caused a large rise in the mean fire size. Percent area burned generally corresponded with fire number data, suggesting that landscapes in interior Alaska were most commonly impacted by fire.

Of the 60 physiographic sections, 53 fell within the study area. Fire statistics stratified by these sections provided a fine scale assessment of fire regimes that clearly illustrated fire activity was highest in interior Alaska. Of the 24 sections in interior Alaska, the fire cycle was under 500 years for 19. Of the remaining five, three sections were very small (less than 405 000 ha). The other two, the Porcupine Plateau and the Yukon-Tanana Uplands had fire cycles over 2000 years and 656 years, respectively. For the remaining physiographic sections outside of interior Alaska, fire cycles were generally over 1000 years. The only exception was the Noatak Lowlands where the fire cycle was 480 years.

Individual large fires had a disproportionate impact on fire regime statistics for several physiographic sections. In 11 sections, one large fire accounted for greater than 50 percent of the area burned over the study period and greatly reduced fire frequency. Two sections, the Kanuti Flats Section and Tozitna-Melozitna Lowland Section illustrated how individual fires can impact fire cycle. Both had the same number of fires, 61 fires/million acres, but a very large fire burned a large percentage of the Kanuti Flats section. As a result fire cycle was 29 years in the Kanuti Flats Section and 260 years in the Tozitna-Melozitna Lowland Section. While the Kanuti Flats section had a shorter fire cycle than the Tozitna-Melozitna Section, it did not necessarily represent a different fire regime since the large fire could have occurred in the Tozitna-Melozitna Section. The calculation of fire regime values stratified by fire weather zones suggested patterns of distribution similar to other spatial analyses with the highest fire activity in interior Alaska.

Discussion & Conclusions

The authors concluded that physiographic sections were the most suitable geographic units for assessing spatial distribution of fire regimes since they accounted for many physiological differences most likely to influence fire regimes (e.g. fuels, soils, topography, and weather).

The study period for this research began well after an aggressive fire suppression policy was instituted in Alaska and several decades following large-scale early settlement of the state. Due to these influences, the natural fire regime could not be calculated. Other fire regime studies that span pre-settlement, settlement, and suppression periods were cited to describe how these activities may have influenced fire regimes but fire history data did not extend back far enough to examine the relative influences of human activity since settlement in Alaska.

Based on the relatively short period of time that had elapsed since settlement and fire suppression policy, this research hypothesized that recent human activity did not have a dramatic influence on the fire regime. Calculated fire cycles for interior Alaska were from 29 through 400 years and where similar to natural fire cycles calculated elsewhere in the U.S. This reinforces the hypothesis that settlement and fire suppression in Alaska did not have a large influence on the fire regime.

Based on differences in vegetation between Alaska and other fire-adapted communities throughout the U.S., some uncertainties surround this hypothesis. Many fire-adapted ecosystems in the contiguous U.S. have high frequency, low severity fires whereas the fire regime in interior Alaska is characterized by low frequency, high severity fires.

Attention is also drawn to the very large fires and the possible connection between their occurrence and weather events. Also noted was that actual fire regime values were higher than those calculated in this study because human-caused fires were excluded.

Reference: Goldstein, G.H. 1981. Ecophysiological and demographic studies of white spruce (*Picea glauca* (Moench)Voss) at treeline in the central Brooks Range of Alaska. Seattle, WA: University of Washington. 193 p. Ph.D. dissertation.

Introduction

A number of experiments were conducted to determine the physiological responses of white spruce to elevation along a gradient at the limit of tree growth in the central Brooks Range, Alaska. This dissertation presented and discussed results of three branches of research: plant community and age structure patterns, environmental relationships of white spruce, and response of plant physiology to environmental variables. Only the plant community and age structure patterns (chapter four) branch of research is reviewed here. The other two branches of research (chapters five through eight) were not pertinent to fire regimes in Alaska. Chapters one through three contained the introduction, literature review, and site descriptions, respectively.

The research reviewed below identified physiological limits of tree growth that were then tested through the other two branches of research. Though this dissertation did not quantify the fire regime, the information from chapter 4 provides some important information on age structure for treeline populations of white spruce and the relevance of fire.

Methods

The study site was located on hillsides to the north of Walker Lake, on the south slopes of the central Brooks Range, Alaska. The study area was at the northern limit of tree growth and forested communities were generally restricted to valley bottoms. The study area spanned an altitudinal gradient of vegetation types from continuous forest at lower elevations to alpine tundra on upper slopes. Where trees were present, the dominant tree species within the study area was white spruce.

Vegetation was quantified at 40 sites along an elevation gradient from 500-1200 m (treeline occurred at 500 – 600 m). Principle component analysis (PCA) and hierarchical clustering analysis was then conducted on site vegetation data to determine how vegetation patterns related to environmental factors and how sites were grouped, respectively. Stand age information was collected by locating four sites at locations with unique environmental characteristics. Two sites were located in continuous forest and two sites were located in scattered trees/alpine tundra (referred to as the combat zone). In each vegetation type one site was located on a south facing slope and a north facing slope. The four sites are referred to as combat zone north (CZN), combat zone south (CZS), continuous forest north (CFN), and continuous forest south (CFS). At each of the four age structure sites, 10 – 15 100 m^2 subplots were established and increment cores and basal cross sections were collected from all trees (including saplings and seedlings). Additionally, growth rates of several trees were measured with dendrometer-bands at each of the four age structure sites.

Results

Treeline was lower on south facing slopes (550 m) relative to north facing slopes (680 m). Hierarchical cluster analysis produced four groups with similar vegetation patterns: (1) riparian forest, north facing continuous forest, and north facing combat zone, (2) south facing continuous forest, and south facing combat zone, (3) alpine tundra on north slopes, and (4) alpine tundra on south slopes. PCA indicated elevation and aspect both influenced vegetation composition.

Age distributions between continuous forest and combat zone sites were different. At the two continuous forest sites age distributions exhibited a negative exponential curve that steeply declined from the sampling date to about 1900 and then leveled off. At the two combat zone sites, recruitment during the first half of the 20th century was underrepresented relative to the continuous forest sites. Additionally, some age classes during the latter half of the 20th century were also underrepresented. Recruitment during the last 20 years was relatively high for north facing sites. Dendrometer-bands indicated that growth rates at the combat zone sites were significantly higher than the continuous forest sites.

Discussion & Conclusions

The lower treeline on southern slopes suggested that moisture stress, induced by greater solar radiation, may be a limiting factor of tree establishment. The moisture difference between north and south facing slopes was further supported by hierarchical clustering analysis which suggested that north facing slopes suffered less from moisture stress since forests occurring on this aspect were classified in the same group as riparian forest. PCA indicated that air temperature, duration of snowpack (both influenced by elevation), and soil moisture (influenced by aspect) were major determinants of floristic composition.

Age structure suggested that in continuous forest stands there were two periods of recruitment: 1775 – 1900 AD and 1900 – 1980 AD. Pre 20th century, recruitment was sporadic, or mortality was high. During the 20th century, recruitment and mortality were constant. The two recruitment periods correspond with two climatic periods (Garfinkel and Brubaker 1980), prior to the 20th century, climate was cool and wet and during the 20th century the climate of interior Alaska has warmed. It was possible that the warmer weather of the 20th century increased tree populations. Age distribution data from the combat zone sites also indicated increased tree populations, however, this trend has been more recent and confined to the latter half of the 20th century. Age data also indicated the treeline has expanded upwards. The above trends may be due to warmer temperatures during the 20th century. At all sites there was no evidence of fire. The absence of fire would be expected based on the uneven age distributions for these stands. This contrasts with treeline stands of black spruce in the Canadian Arctic which were found to be even-aged and regulated by fire (Black 1977).

Reference: Hu, F.S.; Brubaker, L.B.; Anderson, P.M. 1993. A 12 000 year record of vegetation change and soil development from Wien Lake, Central Alaska. Canadian Journal of Botany. 71(9): 1133-1142.

Introduction

This study noted the importance of soil as a driver of boreal forest dynamics, and, based on this observation, investigated the long-term relationship between soils and vegetation to determine if soils were a mechanism of forest change during the Holocene. To study the relationship between soil and vegetation this research quantifies several constituents, describing soils and vegetation for the past 12 000 years, of a sediment core collected from a lake in central Alaska. Fire regimes were qualitatively assessed and were not discussed at length.

Methods

The sediment core was collected from Wein Lake, located west of Fairbanks. Maximum lake depth was 30 m and area was 12 km^2. Vegetation surrounding the lake was primarily closed boreal forest and tree species included black spruce, trembling aspen, paper birch, and white spruce. The understory was primarily ericaceous. Plant macrofossil data and macroscopic charcoal was quantified from samples collected at 10 cm intervals from the sediment core. Pollen, elemental composition, and loss on ignition were also quantified. Radiocarbon dating was used to assign dates along the sediment core. Reported dates are not calibrated.

Results

Four distinct periods of vegetation and charcoal were noted and are listed below. Two measurements of pollen are reported. Relative dominance of species refers to the percentage of the pollen profile occupied by pollen specific to each species. Generally, species whose pollen occupies greater than 20 percent are considered dominant here. The second measurement is pollen accumulation rate (PAR). This refers to the absolute concentration of pollen within the sediment core and is a reflection of the amount of vegetation.

Period I (12 500-10 500 BP). The three most dominant species in the pollen profile were shrub birch, sedges, and willows. Additional species represented included Artemisia and grasses. Macroscopic charcoal was uncommon. PARs during this period were low, suggesting sparse plant cover.

Period II (10 500-9 500 BP). During this time period PARs increased dramatically for many species except sedges. Dominant species included balsam poplar, willows, and shrub birch. Other species represented included ericaceous shrubs, sphagnum moss, and horsetail. Macroscopic charcoal was absent during this time period.

Period III (9500-6500 BP). The beginning of this period was marked by a rapid decrease of balsam poplar and willow pollen coinciding with an increase of white spruce pollen. Approximately 8000 BP alder pollen increased dramatically and became a dominant species in the pollen profile. White spruce pollen was variable during this period with several peaks and troughs. Birch remained dominant in the pollen profile and analysis of birch seeds indicated that paper birch had replaced shrub birch. PARs during this period were fairly low for white spruce and high for paper birch. Macroscopic charcoal is intermittently present.

Period IV (6500-0 BP). At the beginning of this period black spruce pollen first appeared and quickly increased. Both alder and paper birch pollen was less abundant than during the previous zone. Dominant species in the pollen profile included alder, paper birch, and black spruce. Macroscopic charcoal was frequently encountered.

Discussion & Conclusions

The pollen profile during period I suggested the dominant vegetation type was shrub tundra with sparse vegetative cover. Other plant communities in the area around Wein Lake likely included tussock tundra at mesic sites and willow shrublands long lakeshores and on snowbeds. Elemental composition and LOI analysis indicated soil erosion into Wein Lake was intense. During period II, the pollen profile indicated a shift to dense stands of balsam poplar with a willow understory and extensive thickets of dwarf birch. High concentrations of Sphagnum indicated bryophytes also expanded. Elemental composition and LOI results suggested forest development had a profound effect on soils; both acidity and organic content increased. Soils likely changed from entisols to inceptisols during this period. Results from elemental composition analysis suggested the aquatic environment was also affected; both nutrient loading and algal community productivity increased. During period III forest type shifted from balsam poplar to white spruce-paper birch woodlands and shrub birch tundra disappeared. The sporadic occurrence of macroscopic charcoal indicated fire frequency increased. During period IV pollen data indicated closed spruce forest dominated by black spruce developed around the lake. Macroscopic charcoal indicated fire frequency was highest during this period.

Soils data suggested that the vegetation shift 9500 BP (i.e. transition from woodlands to closed black spruce forest) had no impact on soils. This contradicted previous forest succession (Van cleve et al. 1983, Van Cleve et al. 1986, Van cleve et al. 1991) and paleoecology (Engstrom and Hansen 1985) research which suggested succession from white spruce to black spruce is accompanied by the development of anaerobic waterlogged soils. One explanation for this contradiction was that the shift from white to black spruce was not caused by soil paludification as indicated by the lack of increased in sphagnum moss spore counts, which would be expected during soil paludification. An alternative explanation was that more recent processes within the watershed removed elemental composition patterns that would have indicated soil paludification.

Results suggested that balsam poplar forests coverage during the early Holocene was more extensive than thought. However this research did not shed much light on why balsam poplar transitioned to white spruce. Three possible explanations are given: succession to white spruce, climate change that favored white spruce, and lack of a suitable seedbed (i.e. exposed mineral soil) for balsam poplar establishment. Succession was unlikely because several millennia passed before balsam poplar transitioned to white spruce, and climate change was unlikely because 9500 BP, the climate changed to favor balsam poplar rather than white spruce. The third possibility was most likely given that elemental composition analysis indicated that exposed mineral soil decreased during period II.

Results from this and other research (Anderson et al. 1990) suggested the decline of white spruce and increase of black spruce may have been separate events. If this were the case, it would explain why soil characteristics failed to indicate paludification. Climatic cooling, rather than soil paludification may have caused the change in forest type. The increase in fire activity may have been caused the arrival of black spruce which is a highly flammable fuel type (Payette 1992).

Reference: Hu, F.S.; Brubaker, L.B.; Anderson, P.M. 1996. Boreal ecosystem development in the northwestern Alaska range since 11 000 BP. Quaternary Research. 45(2): 188-201.

Introduction

Numerous studies (Ritchie 1985, Macdonald 1987, Ritchie 1987, Bonan et al. 1990, Hu et al. 1993, Anderson and Brubaker 1994, Earle et al. 1996) have suggested the mechanism for historic vegetation shifts were changes in soils and fire regimes induced by climate change. The purpose of this study was to assess these relationships at a site west of the Alaska Range and describe the history of vegetation and limnological environment of the lake for the past 11 000 years.

Methods

The study area, Farewell Lake, was located near the southwest corner of the Alaska Range, approximately 3 km west of the South Fork of the Kuskokwim River. The depth of the lake is 55 m and the area is 4 km^2. Forest types surrounding the lake include black spruce muskeg on poorly drained lowlands and mixed boreal forests on well drained uplands. One sediment core was collected from the lake. Analyses were conducted to identify pollen, macroscopic charcoal, macrofossils, geochemistry, and magnetic susceptibility. Radiocarbon dating was used to classify dates within the sediment core. Reported dates are uncalibrated.

Results

Vegetation (pollen and plant macrofossil data) and fire regimes (macroscopic charcoal data) indicated three distinct periods during the Holocene.

Period I (11 000-8 000 BP). The pollen profile during this period was dominated by birch, balsam poplar, willows, and grasses. Other species well represented in the pollen profile included Artemisia, juniper, rose, and horsetail. PARs were lower than the other two periods. Mollusks were most abundant and diverse. Charcoal was abundant until 8 500 BP.

Period II (8000-4000 BP). During this period, pollen percentages for dominant species were unstable. Dominant species (and general trends in pollen percentage) included white spruce (increasing), black spruce (increasing after 6000 BP), birch (decreasing), and alder (increasing). Other notable changes in vegetation included the appearance of paper birch seeds, the appearance and higher pollen percentages of white spruce over black spruce. Macroscopic charcoal fragments were present but occur irregularly. Results indicated algae were common in the lake and mollusks representation was relatively poor.

Period III (4000-0 BP). This period was distinguished by consistently higher percentages of black spruce over white spruce pollen and higher percentages of sphagnum spores. Birch decreased slightly, while willows and alders increased slightly. PARs were highest during this period. Mollusks became more diverse once again, but were still less diverse than during period I. Macroscopic charcoal was most abundant during this time period.

Discussion & Conclusions

For period I, the landscape was likely characterized by extensive areas of birch shrub tundra. Less common plant communities included balsam poplar forest in riparian areas and Artemisia-juniper shrublands on well drained south facing slopes. The occurrence of balsam poplar and some mollusk species indicated the summer climate during this period was warmer than present. The relatively high frequency of fires may have been responsible for the intense soil erosion indicated by elemental composition analysis.

Though period II was marked by a number of vegetation changes, a forest-tundra ecotone probably occupied the majority of the area around Farewell Lake until 6000 BP when it was eclipsed by closed boreal forest. About this time, paper birch, alder, and black spruce also became important species within the pollen profile, with paper birch and white spruce growing on well drained upland sites and alder growing in thickets on mountain slopes and riparian areas. Changes in mollusk characteristics and elemental composition suggested the climate became cooler. Macroscopic charcoal results indicated that fires were relatively rare during period II. The shift from white to black spruce approximately 4000 BP marked the beginning of modern boreal forest in the region. Forest composition during this period included mixed boreal forest at well drained upland sites and black spruce muskeg at poorly drained lowland sites. Black spruce forest density probably reached current levels approximately 1200 BP when black spruce PARs reached current levels. Increased erosion indicated by higher sediment deposition may have been caused by fires which were more frequent during period III, than either of the previous two periods. Glacial advances in the northern Alaska Range during this period (Calkin 1988) suggested the climate became cooler and wetter. Advancing glaciers may have also created more riparian habitat, explaining the increased abundance of willow, alder and poplar pollen. Elemental composition suggested soils became paludified during the transition from periods II to III, though this conclusion was equivocal.

The authors of this study hypothesize that climatic cooling, through a cascading series of events leading to soil paludification, was the mechanism that shifted the dominant tree species from white to black spruce, which is better adapted to nutrient poor, waterlogged soils (van Cleve et al. 1986, Viereck et al. 1992). This hypothesis agreed with previous assessments of the mechanisms of white to black spruce forest succession in interior Alaska (van cleve et al. 1983, van Cleve et al. 1986, Van cleve et al. 1991). The higher frequency of fires was likely caused by the increasing coverage of black spruce forest which is highly flammable and prone to stand-replacing fires (Dyrness et al. 1986). The increasing frequency of fires would have perpetuated and expanded black spruce coverage, as they are well adapted to fires. This research and two other studies (Hu et al. 1993, Earle et al. 1996) conducted in interior Alaska concluded that the late-Holocene increase in fire frequency was coincident with the expansion of black spruce. This is notable given the different charcoal analysis methodologies among each study and because these changes occurred at different times at each study site. The shift towards higher fire frequency and expansion of black spruce has been noted by several paleoecology studies in Canada. However, some believe, as this study does, that soil paludification, induced by climate change, was responsible for the shift (Payette 1992) and others attributed the shift to soil paludification induced by succession (Lamb 1980, Engstrom and Hansen 1985, Liu 1990).

Reference: Kasischke, E.S.; Williams, S.D.; Barry, D. 2002. Analysis of the patterns of large fires in the boreal forest region of Alaska. International Journal of Wildland Fire. 11(2): 131-144.

Introduction

Links between fire, climate, and vegetation were explored based on fire perimeter data in Alaska. The goal of this research was to identify temporal and spatial patterns of fire across boreal forests for the last half of the 20th century.

Methods

Sources of fire data used to quantify fire regimes included total seasonal area burned estimates for the period 1960-2001 from Alaska Fire Service (AFS), interseasonal area burned estimates for the period 1994-2001 maintained by the National Interagency Coordination Center (NICC), and fire data for the period 1950-99 from the large fire database (LFDB). Three components of interseasonal fire patterns were assessed: variation for decadal average area burned, distribution of fire size between large and small fire years, and temporal distribution of fires during the burn season for large and small fire years. Large fire years were defined as those with an annual area burned that exceeded the long term average by a factor of 1.5; all others were considered small fire years. Spatial fire patterns were evaluated by calculating fire cycle for areas stratified by biological, topographical, and climate-based parameters derived from pre-existing geographic datasets. Potential sources of error included probable underestimates of area burned during the 1950's and 60's due to inaccurate fire perimeters and the omission of fires less than 400 ha (for most years) from the LFDB. Omission of small fires was normally distributed across nearly all years and any bias was likely minor.

Results

Correlation of AFS and LFDB seasonal area burned statistics was reasonably good for 80 percent of the years for the period 1950-99. For the remaining 20 percent of the period (10 years) correlation was poor and was distributed throughout the 1950's (4 years), 1960's (3 years), and, 1970's (3 years). Relative to AFS seasonal area burned statistics; the LFDB data was higher for six years and lower for four years.

Analysis of annual area burned data suggested that for the period 1960-99, 73 percent of the area burned during the ten most active fire seasons. There were no temporal trends and the major determinant of decadal area burned was the frequency of large fire years. Average number and size of fires was higher during large fire years.

NICC data indicated the length of the fire season is longer for large fire years. During small fire years the majority of the area burns in June and July while during large fire years the majority of the burning occurs later, primarily in July. The fire season also extends later in the season during large fire years. Pre-existing climate and fire research (Hess et al. 2001) has correlated large fire years with positive phase El Nino events which are associated with warmer and drier summers.

Analysis of aspect in relation to area burned suggested differences in area burned among different aspects. North facing slopes contained 44 percent of the area burned while south facing slopes contained 35.5 percent. This may be due to unique fire regimes of vegetation types specific to north and south facing slopes.

Analysis of all fire perimeter data for the state of Alaska indicated that 96 percent of fires in Alaska occur within the 11 ecoregions that represent interior Alaska. Further spatial analysis focused on ecoregions of interior Alaska. Five environmental traits were averaged for each ecoregion and ecoregions were then assessed for relationships between these parameters and fire cycle. Fire frequency had positive relationships with percent tree cover ($r = 0.65$ and $P < 0.06$) and average growing season temperature ($r = 0.63$ and $P < 0.06$). The relationship was negative for growing season precipitation ($r = 0.61$ and $P < 0.06$). No significant relationship was observed for average lightening strike frequency and average elevation but subtle trends were noted including a slight positive response of fire frequency to lightening ($r = 0.31$ and $P \geq 0.06$) and a slight negative response of fire frequency to elevation ($r = 0.44$ and $P \geq 0.06$).

Comparison of fire location coordinates from the LFDB and elevation for interior Alaska (minus the two coastal ecoregions which have little topographical relief) suggested fire cycle decreased with elevation until 200-400 m, at which point it increased. The pattern above 200-400 m was attributed to slower growth rates at higher elevations which gradually increased the time regenerating forests (and tundra above elevations of 600-700 m) accumulated enough fuels to become flammable. Below 200-400 m, higher fire cycle was attributed to the higher concentration of fire resistant wetlands and riparian forest.

Discussion & Conclusions

The data source for this research was similar to past studies of fire patterns in Alaska (Barney 1971, Gabriel and Tande 1983), but an additional 20-30 years of data provided for a robust analysis of trends and the availability of climate, topography and vegetation datasets afforded the opportunity to assess the relative influences of environmental factors. This research generally agrees with conclusions from other studies of fire patterns in boreal forests (Skinner et al. 1999, Murphy et al. 2000, Stocks et al. 2002). Analysis of fire frequency throughout Alaska suggested that while large fires occured across the state their frequency outside of interior Alaska was dramatically lower. This conclusion agreed with past assessments of fire frequency in the state (Gabriel and Tande 1983). Correlations between fire frequency and climate were also similar to previous research for lightning (Reap 1991) and seasonal precipitation (Hess et al. 2001). The overall pattern of fire frequency for interior Alaska observed in this study increased on a west to east gradient. This may be due to regional temperature and precipitation gradients.

Reference: Lynch, J.A.; Clark, J.S.; Bigelow, N.H.; Edwards, M.E.; Finney, B.P. 2002. Geographic and temporal variations in fire history in boreal ecosystems of Alaska. Journal of Geophysical Research. 107: 8152, doi: 10.1029/2001JD000332, [printed 108(D1), 2003].

Introduction

Charcoal and pollen analysis were conducted on lake sediment cores collected from five lakes in Alaska. Objectives included assessing the influences of early settlement and regional differences in climate on fire regimes, and determining the relative influence of climate and vegetation on patterns of fire during the Holocene.

Methods

Of the five lakes sampled, two, Dune and Deuce lakes, were located in interior Alaska. Dune Lake is located 71 km southwest of Nenana and Deuce Lake is located adjacent to the University of Alaska Fairbanks, Fairbanks, Alaska. Rock, Portage, and Arrow lakes, were located on the Kenai National Wildlife Refuge (NWR), Alaska. All lakes are small (surface area less than 12 ha) and relatively shallow (maximum depth less than 16 m). Charcoal from lake sediment cores was used to determine CHARs for each lake and *mFRI* at Dune Lake. Historic vegetation composition was reconstructed from pollen data and previous paleoecological research (Bigelow 1997). Climate reconstruction was based on previous research (Ager 1971, Anderson and Brubaker 1994, Edwards and Barker 1994, Bigelow 1997, Hu et al. 1998, Abbott et al. 2000, Finney et al. 2000). To calculate *mFRI*, spikes in charcoal representing local fires were identified by setting threshold conditions based on knowledge of charcoal accumulation in sediments. Present day forest is primarily black spruce and paper birch at the Kenai NWR sites; white spruce, paper birch, and quaking aspen at Dune Lake; and black spruce, white spruce, paper birch, and quaking aspen at Deuce Lake. Samples dates were assessed by AMS (Accelerator Mass Spectrometry) ^{14}C dating and dates were converted to calibrated years before present.

Results

CHARs were highest at Dune Lake and lowest at Rock and Deuce lakes. Arrow and Portage lakes had intermediate CHARs. At all five lakes, CHARs were highest after *ca.* 1850.

Pollen records from the lake sediment cores were consistent from 1000-0 BP. Dominant species recorded in the pollen record at Arrow, Portage and Dune Lakes were black spruce, alder, birch, and to a lesser extent, poplar. Small changes in the pollen record during this time period included increased white spruce and alder pollen, decreased birch pollen at Rock Lake after a fire in 1947, and increased black spruce and alder pollen at Portage Lake.

Prior to 5500 BP, data from the Dune Lake sediment core indicated low CHARs and a lack of distinct charcoal peaks. The pollen record from Dune Lake indicated that vegetation during this period was dominated by birch, alder, and white spruce. After 5500 BP, CHARs increased appreciably and charcoal peaks occur, on average, every 127 years. These changes in the fire regime coincide with a sharp increase in black spruce pollen *ca.* 5800 BP. The fire regime at Dune Lake for the period 5500-0 BP was stratified by *mFRI*. For the first sub-period, 5500-2400 BP, the *mFRI* was 97 years. For the second sub-period, 2400-0 BP, the *mFRI* was 198 years. Pollen data indicated vegetation composition was stable for the entire period (5500-0 BP).

When the Weibull model was fitted to each sub-period the shape parameter was greater than one (1.6 for 5500-2400 BP and 2.3 for 2400-0 BP). In both cases a likelihood ratio test rejected the hypothesis that the shape parameter could be equal to one indicating that hazard of burning increases with time since last fire.

Discussion & Conclusions

The higher CHARs in interior Alaska were consistent with current knowledge that fire frequency in interior Alaska is higher than on the Kenai Peninsula. The charcoal record from the Kenai Peninsula indicated large stand replacing fires were more frequent following early settlement. Since human ignitions were responsible for a number of recent fires on the Kenai Peninsula, including a large fire in 1947 (De Volder 1999), it is reasonable to assume the fire regime shift was caused by humans. However, the influence of settlement was possibly augmented by a coincident warming trend (Chapman and Walsh 1993, Overpeck et al. 1997) which may have dried fuels and encouraged larger fires. There was little to no apparent influence of early settlement on fire frequency in interior Alaska: CHARs increased slightly and there was no change in *mFRI*.

The dramatic and sustained increase in fire frequency at Dune Lake *ca.* 5500 BP occurred during a transition from mixed boreal forest to black spruce. Data indicated that fire was not an important ecological process in the mixed boreal forests. However this type of analysis only detects higher intensity, stand-replacing fires and low-intensity, surface-fires may have been missed. Climate does not explain low fire frequency during this period. The climate was warmer and drier than present which would suggest higher fire frequency. One explanation for lower than expected fire frequency is the lower flammability of the mixed boreal forest (Brown and Davis 1973, Van Wagner 1983, Amiro et al. 2001, Hely et al. 2001). Higher fire frequency after 5500 BP also cannot be explained by climate which became cooler and wetter *ca.* 7000 BP. Again, fuel type was believed to be the most plausible cause of higher fire frequency since black spruce forests are known to support relatively frequent crown fires (Viereck 1973).

Approximately 2400 BP, the fire regime shifted again and marked the beginning of the present day fire regime. This shift was attributed to climate. Vegetation did not change appreciably and the climate became even cooler and wetter. The change in climate would have reduced lightning storms and increased fuel moisture; both explain the reduction in fire frequency.

CHAR data did not indicate a large difference between fire regimes on the Kenai Peninsula and interior Alaska despite differences in regional climate. CHAR data suggested that early settlement increased fire frequency on the Kenai Peninsula but not in interior Alaska. Long-term fire frequency data from Dune Lake indicates that both climate and vegetation composition influence the fire regime of boreal forests in interior Alaska.

Reference: Lynch, J.A.; Hollis, J.L.; Hu, F.S. 2004. Climatic and landscape controls of the boreal forest fire regime: Holocene records from Alaska. Journal of Ecology. 92(3): 477-489.

Introduction

High resolution analysis of charcoal, pollen, macrofossil, and lithological characteristics was conducted on lake sediment cores collected from two lakes in Alaska. The purpose of this research was to reconstruct vegetation, *mFRI* and effective moisture for the past 7000 years, and provide a better understanding of how climate and vegetation influence fire regimes.

Methods

Two sampling sites, Chokasna and Moose Lakes, were located in Wrangell-St. Elias National Park and Preserve in southcentral Alaska. The lakes are located 20 km apart, and are small (less than 25 ha in surface area) with a relatively shallow depth (maximum depth is less than 5 m). Gently rolling hills surrounding Moose Lake and support mixed boreal forest. The forest is characterized by an overstory of paper birch, trembling aspen, and white spruce and an understory of dense willow and alder cover. At Chokasna Lake, the landscape is flat and supports black spruce forest. Fossil pollen was used to reconstruct vegetation. Fire regimes were quantitatively assessed by calculating CHAR and *mFRI*. Macrofossil and sediment characteristics were used to estimate water level depth, which was in turn interpreted to infer local climate. Samples dates were assessed by AMS (accelerator mass spectrometry) ^{14}C dating and dates were converted to calibrated years before present

Results

The pollen record at Moose and Chokasna lakes was divided into three periods (table 12). At Moose Lake, prior to 6500 BP, the pollen profile was dominated by sedges, spruce and birch. From 6500-2400 BP, birch and alder pollen were dominant while black and white spruce pollen percentages were low. In the most recent time period, 2400-0 BP, black spruce was dominant until *ca.* 1800 BP, when percentages declined and alder became the dominant plant represented in the pollen profile. At Chokasna Lake, sedges, and white and black spruce were dominant in the pollen profile for the period 7100-5100 BP. From 5100-2000 BP, alder and birch pollen percentages increase. From 2000-500 BP, black spruce is dominant and white spruce pollen percentages decrease.

The pollen data suggests that a closed boreal forest dominated by white spruce, black spruce, and paper birch existed at both lakes during the past 7000 years. Black spruce replaced white spruce as the dominant species *ca.* 2400 BP at Moose Lake and *ca.* 2000 BP at Chokasna Lake. However the transition was more pronounced at Chokasna Lake where mixed boreal forest changed to black spruce, whereas at Moose Lake mixed forest remained. These different vegetation transitions were attributed to distinct topography at each site. The poorly drained lowlands surrounding Chokasna Lake would be more likely to become anaerobic as effective moisture increased from 3800-1600 BP. Black spruce would have outcompeted mixed boreal forest species on these soils.

Both lakes share a similar climate history that roughly mirrors vegetation. Prior to 7200 BP effective moisture was lowest and the climate was likely very dry. For the period 7200-3800 BP effective moisture was higher, but generally less than present. For the period 3800-1600 BP effective moisture progressively increased.

CHARs at Moose Lake were highest for the periods 6500-6000, 5800-5000, *ca.* 4400 and after 3800 BP. With the exception of a peak at *ca.* 4000 BP, rates were very low from 5000-3800 BP. CHARs at Chokasna Lake were highest from 6800-6600, 5600-3600, and 3100-500 BP. CHARs were lowest from 6600-5600 BP and, with the exception of a peak *ca.* 3600 BP, from 4500-3100 BP.

There are four periods of distinct *mFRI*s for Moose and Chokasna lakes (table 12). At Moose Lake, fires were most frequent for the periods 5800-5000 and 3800-0 BP. At Chokasna Lake, fire frequency was highest for the periods 5600-4600 and from 3100-0 BP. These periods of high fire frequency correlate with periods of cooler and wetter climate. At both sites, *mFRI* is at the upper limit or higher than previously reported *mFRI* values for the vegetation types represented; this may be due to low sample resolution which would not detect fires that occurred within 40 years of each other. There was an important difference in fire history between the two lakes during the late Holocene. For the period 2000-0 BP *mFRI* was higher at Moose Lake and from 3100-2000 BP *mFRI* was higher at Chokasna Lake.

Discussion & Conclusions

The positive correlation between fire frequency and effective moisture is counterintuitive, but consistent with previous research in Alaska and parts of Canada. Carcaillet et al. (2001) proposed that increasing moisture *ca.* 3000 BP was primarily limited to the winter months and summers were drier creating more favorable conditions for ignition and fire spread. This explanation is less applicable to southcentral Alaska were historic weather data indicates severe fire seasons are most correlated with less winter precipitation (Hess et al. 2001). Likewise, a hypothesis explaining that shifts to more flammable fuel types increased fire frequency (Lynch et al. 2002) are not applicable because shifts in fuel type at these sites occurred after fire frequency changed. Another explanation is that changes in atmospheric circulation increased lightning frequency in the region which in turn increased fire frequency.

Given the similar climate between both sites, site flammability provides a more reasonable explanation for the lower *mFRI* at Chokasna Lake after 2000 BP. The black spruce (prevalent at Chokasna Lake) is more prone to fire than mixed boreal forest (found at Moose Lake). This supports the lower *mFRI* measured at Chokasna Lake.

These results indicate that warmer and drier weather does not necessarily translate into higher fire frequency. These results are also inconsistent with current predictions of a more active fire regime associated with future climate change scenarios. This research indicates that a warming climate in southcentral Alaska may reduce fire frequency through reduced lightning frequency and increase carbon storage through expansion of late-succession spruce forests which insulate permafrost. This could maintain or increase current rates of carbon storage. The interactions among climate, vegetation and fire are complex and more research is needed to understand how future climate change scenarios will influence fire regimes and carbon storage in southcentral Alaska.

Reference: Mann, D.H.; Fastie, C.L.; Rowland E.L.; Bigelow, N.H. 1995. Spruce succession, disturbance, and geomorphology on the Tanana River floodplain, Alaska. Ecoscience. 2(2):184-199.

Introduction

The Drury Hypothesis (Drury 1956) has been a tenet of ecology in Alaska for several decades. It states stands of riparian white spruce are a mid-seral stage that transition to black spruce. The proposed mechanism for this transition is a reduction in site quality, caused by accumulating organic material and decreasing active layer thickness, best tolerated by black spruce (Bonan and Korzuhin 1989). Though this hypothesis was supported by previous literature (Viereck 1970, Van cleve et al. 1991, Viereck et al. 1993), a host of research has presented evidence that contradicts the Drury hypothesis (Dirschl 1972, Gill 1972, 1973, Nanson and Beach 1977). The primary objective of this research was to test this hypothesis by evaluating disturbance regimes and succession on a chronosequence of geological landforms at a site on the Tanana River floodplain.

Methods

The study area was located on the Tanana River floodplain near Fairbanks, Alaska. Forest types included closed black and/or white spruce, black spruce-tamarack woodlands, and open mixed stands of white spruce and paper birch. Forests are interspersed with shrub, herbaceous, and graminoid peatlands.

Two age types were described in this research: surface age refers to the date the landform was created, and stand age refers to the date the overstory established. Landforms were dated using a combination of radiocarbon dating and aging relative to position of adjacent landforms. Forest stands were aged by collecting increment cores to assess stand age and basal cross sections, when fire scars were encountered, to determine fire dates. Increment cores were not cross dated to account for missed piths, height above root crown, or missing growth rings. Basal cross sections were cross-dated to accurately estimate fire year. A time-since-last-fire (TSLF) map was prepared for the study area based on age data, fire scar locations, and stand delineations from aerial photographs. Vegetation composition and soils were assessed with plot data. The relationships among spruce species, organic layer thickness, and active layer depth postulated by the Drury hypothesis was assessed by establishing transects across gradients of different spruce forest types.

Results

Four fluvial landforms, surfaces A-D (oldest to youngest) were identified: surface A established at least 3000 ^{14}C BP, surface C established between 2750 ^{14}C BP (± 50 years), surface B formed sometime after surface A but before surface C, and most parts of surface D dated from 700-100 ^{14}C BP. Superimposed on these surfaces were two geomorphic units: the meander belt and the backswamp. The meander belt consisted of active and abandoned river channels, naturally formed levees, and sediment ridges. The backswamp section of the floodplain was located between the meander belt and the uplands. Drainage restricted by the meander belt prevented water from moving from the backswamp to active river channels. The backswamp was primarily located on surfaces A and B and the meander belt was primarily located on surfaces C and D. Forest type was nearly pure black spruce on surfaces A and B, white spruce on surface D, and a mosaic of white and/or black spruce on surface C.

Surfaces A and B had two age cohorts, dated to 1850 and 1920. The same age cohorts occurred on surface C, as well as a third cohort dated to at least 1810. Surface D had five cohorts dated to 1530-1595, *ca.* 1660, *ca.* 1770, *ca.* 1840, and 1920. Tree age distributions and fire scars indicted a fire burned through nearly all of surfaces A and B and parts of surfaces C and D in 1910. A second fire burned the entire study area, except for surface D, in 1816 leaving few surviving trees. Fire scars supported by cohorts indicate two additional fires in 1793-94 and 1594-95. Probable fire scars unassociated with age cohorts were also observed. The number of trees surviving the fires in 1816 and 1910 was higher in stands of white spruce relative to black spruce.

Analysis of soil and vegetation data from the spruce gradient transects revealed no statistically significant correlation between active layer depth and frequency of black or white spruce. There was a weak correlation between black spruce frequency and organic layer thickness at some locations. Correlation between active layer depth and organic layer thickness ranges from significantly positive to weakly negative.

Discussion & Conclusions

The authors concluded that primary floodplain disturbances were flooding and fire. This was contrary to previous research which stated that fire was not an important disturbance agent on the Tanana floodplain (Van cleve et al. 1991). Results indicated the *mFRI* at this site was 70 -100 years. Significant flooding (sediment deposition > 2 cm) occurred every 100 years on the meander belt and every 500-1000 years in the backswamp.

Results of this research did not support the Drury hypothesis. Expected correlations between organic layer thickness and active layer depth, and between organic layer thickness and black spruce frequency did not conform to the Drury Hypothesis.

The backswamp may support a unique floodplain environment rather than representing the climax seral stage as hypothesized by Drury (1956). Results suggested soils conditions that support black spruce were a result of geomorphology rather than the climax stage forest succession. Additionally the backswamp area may support a more active fire regime than other riparian areas because the black spruce-ericaceous shrub forest type inhabiting the backswamp is generally assumed to be more flammable relative to riparian white spruce forest. White spruce has persisted on meander belts for millennia without transitioning to black spruce; further indicating the Drury hypothesis was false. Three natural processes allow white spruce to persist by suppressing permafrost aggradation and peat accumulation that would otherwise favor black spruce: fires, flooding, and groundwater intrusion. Though fires occur with equal or higher frequency in the backswamp; it is unlikely they reduce permafrost because the high water table minimizes consumption of the organic mat that insulates permafrost from solar heating. Additionally flooding and groundwater intrusion in the backswamp are minor.

Reference: Mann, D.H.; Plug, L.J. 1999. Vegetation and soil development at an upland taiga site, Alaska. Ecoscience. 6(2): 272-285.

Introduction

Fire initiated secondary succession in North American boreal forest appears to have a large effect on landscape-scale distribution of vegetation (Rowe 1961, Zackrisson 1977, Foster 1983, Johnson 1992). With this in mind, the authors investigated the influence of underlying surface material age on vegetation distribution on a floodplain in interior Alaska. Specific objectives included describing how vegetation and soils change along the geologic chronosequence and advancing a hypothesis to explain long-term ecosystem development.

Methods

The study area was located at the junction of Riley Creek and the Nenana River, Alaska. Vegetation within the study site was generally mixed boreal forest and stands were dominated by trembling aspen, white spruce, or balsam poplar.

The study site included a number of glacially deposited alluvial fans and terraces deposited by Riley Creek. Two alluvial fans (surface VIII and VII, respectively) were dated to 12 300 BP (± 500 years) and 10 000 BP (± 500 years) (Wahrhaftig 1958, Ten brink and Waythomas 1985, Hamilton 1994). Dates for the six other surfaces (all terraces deposited by Riley Creek) were estimated based on rates of downcutting and loess deposition. The estimated mean establishment dates (and range) for surfaces VI – II were: 8000 BP (9000-6300 BP), 6200 BP (8700-3500 BP), 4700 BP (6200-3000 BP), 1600 BP (2200-900 BP), and 300 BP (600-0 BP), respectively. Surface I was the modern floodplain for Riley Creek.

Study sites were established on each of the eight geomorphic surfaces where vegetation composition, stand age, fire history, and soils were assessed. Stand age and fire history data included increment cores collected 15-40 cm above the root crown from randomly selected trees and partial basal cross sections collected from trees with fire scars. Clustering analysis and ordination were used to analyze patterns of vegetation and relationships with environmental parameters, respectively.

Results

The geomorphic surface was broadly separated into two categories: surfaces I and II were affected annually by flooding from Riley Creek and neither had evidence of fire. In contrast, surfaces III-VIII were unaffected by flooding and evidence of fire was common. The dominant vegetation on surfaces I and II included: alders, willows, herbs, balsam poplar (seedlings and saplings), and white spruce (seedlings). Vegetation on surfaces III-VIII was characterized by forest. Dominant trees varied by site, but included balsam poplar and white spruce at sites closest to the creek, and trembling aspen at sites farthest from the creek. The understory included feathermosses and ericaceous shrubs. Ericaceous shrubs and trembling aspen were absent on surface III. Balsam poplar was uncommon on surfaces V-VIII.

Of 15 environmental variables measured, eight were not significantly correlated with geomorphic surface age. Two variables, temperature and mean organic layer thickness, showed weak positive correlations with surface age. Four variables, had statistically significant correlations with surface age: soil particle size became smaller with increasing surface age as loess accumulated over sand and/or gravel, soil horizons (Bw and BC) became thicker as surface age increased, root sprouting plant cover increased with surface age, and tree species composition shifted from surfaces I-IV, but remained stable from surfaces V-VIII. Tree recruitment and fire scar data suggested fires occurred in the 1770's, 1810's, 1872, and 1924. Preceding 1920, the approximate date fire suppression policy was instituted, *mFRI* for the study area was 40-60 years. A number of fires occurred within close proximity to the study area after 1920 but were quickly extinguished.

Discussion & Conclusions

Long-term vegetation chronosequence suggest that transere succession, defined as the cumulative influence of primary succession and secondary succession disturbance regime on ecosystem development, on well drained fluvial sites produced fire adapted vegetation dominated by trembling aspen and ericaceous shrubs. Ordination of vegetation and environmental variables suggested that transere succession reached an endpoint at surface V. Vegetation composition among the four younger surfaces indicated patterns of succession from primarily herbs and tall shrubs developing at surfaces I and II, to forest by surface III, and then transitioning from riparian forest to fire adapted forest from surface III to V. Vegetation composition for surfaces V-VIII was stable and soils data were stable for a longer period of time on surfaces III-VIII. Results indicated approximately 4000 years passed before vegetation adapted to the fire regime; paleoecological research has reinforced this conclusion. Similar time periods were required for boreal vegetation types to reach equilibrium following the last glacial period in the Hudson Bay lowlands (Klinger and Short 1996) and interior Alaska (Hu et al. 1996).

In regards to secondary succession, these results challenge previous work describing succession in white spruce (Viereck et al. 1986), which asserted post-fire succession in white spruce lasted 200-250 years and proceeded through a hardwood stage of trembling aspen and paper birch, ultimately replaced by shade tolerant white spruce in the understory. Results of the Riley Creek research suggested hardwood stands were not a precursor to white spruce, but rather, were in equilibrium with the natural fire regime. This conclusion agreed with other research conducted on fire regimes in Alaskan hardwood stands (Yarie 1981).

Fire was the proposed mechanism for transere succession and the authors used the term “fire hardening” to explain the transition from fire intolerant, to fire tolerant species. Essentially, younger stands (surfaces I and II) had been exposed to few, if any fires. As millennia passed, older sites were impacted by an increasing number of fires. With each passing fire, plants adapted to post-fire regeneration were selected for, thus older sites had a higher percentage of plants able to resprout following fire. The only exception to this pattern was balsam poplar. Despite its ability to rapidly resprout after fire, damage from winter moose browsing may have restricted it to near river sites.

Fire-adapted species on older surfaces were likely self perpetuating so long as fires occurred relatively frequently. The fuel type of this forest supported low-intensity surface fires that favored re-sprouting from surviving underground plant parts. Warm soil temperature and high pH litterfall inhibited factors (e.g. cold soil temperatures, low decomposition, and peat buildup) conducive to soil paludification and subsequent transition to coniferous forest.

Reference: Quirk, W.A.; Sykes, D.J. 1971. White spruce stringers in a fire-patterned landscape in interior Alaska. In: Slaughter, C.W.; Barney, R.J.; Hansen, G.M., eds. Fire in the Northern Environment. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station: 179-197.

Introduction

Long narrow bands of white spruce forest embedded within larger stands of black spruce, known as spruce stringers, are common in the boreal forest of interior Alaska. These are possibly fire refugia having survived successive wildfires. Surrounding forest was primarily composed of a heterogeneous mixture of white spruce, quaking aspen, and paper birch. Tree age data from spruce stringers and surrounding stands were analyzed to determine the disturbance history of the area. Comparative stand and soils data were also collected from the stringer and adjacent forests.

Methods

The study site was located on a south facing slope in the Caribou-Poker Creeks Research Watershed 48 km north of Fairbanks, AK. The site included six spruce stringers. Data were collected along a line transect at 640 m above sea level that intersected all six stringers as well as the surrounding forest. Along the transect cores were collected from living trees and cross sections were collected from fire-killed trees. Ring counts from both types of samples were assumed to represent the age of the tree. In some instances tree age was estimated based on diameter. Additional measurements included tree density, overstory species composition, tree height, soil profiles, soil temperature, soil moisture, and descriptions of vegetation.

Results

Many stand measures had similar values among all six stringers including age of dominant trees and composition and density of the understory and overstory. Those that did not included tree density which varied among the spruce stringers and ranged from 298-913 trees/ha. Among the adjacent forest, stand composition was more heterogeneous but tree age was consistent. In both, spruce stringers and adjacent forest, age distribution appeared to cover several cohorts. Spruce stringers appeared to have a dominant overstory of white spruce and paper birch that regenerated from the same disturbance event and an uneven-aged understory of white spruce. Age distribution in the adjacent forest was similar except the overstory was younger and composed primarily of paper birch.

Age estimates for dominant trees in the adjacent forest had a range of 30-38 years. Thus indicating the area burned around 1930. The oldest fire-killed tree from the adjacent forest was 65 years old suggesting that the previous fire occurred in 1865. In the spruce stringers, stand age data suggested a different fire regime. The dominant trees had an average age of 193 years old and the oldest tree was 208 years old. Thus, these small stands had survived both of the previous fires recorded in the adjacent forest and had not burned for at least 200 years. The stand was not examined for fire scars but there was little physical evidence of past fires with the exception of a few old rotting logs on the forest floor.

There was no evidence to identify whether the spruce stringers are relicts of once more extensive stands of mature white spruce of if they were unique stands. However, it appeared that in the adjacent forest frequent fire was maintaining an early seral stage of paper birch that would burn before white spruce in the understory could become dominant.

An investigation of the topography and soil profile indicated that spruce stringers were located in unique terrain. Each stringer was located below an escarpment and within small depressions. The upslope end of each stringer, near the escarpment, was also the source of a small spring or seep. The soil in the spruce stringers had a thicker organic layer relative to the adjacent forest and as a result had a higher water holding capacity. Soil temperature readings suggested that soils in the adjacent forest (6 ºC) were two degrees warmer than soils in the spruce stringers (4 ºC). The lower soil temperature in the spruce stringers was likely an artifact of the thicker organic layer which had better insulating properties. Soil moisture in the spruce stringers (35-37 percent) was higher than in the adjacent forest (23-24 percent). The springs, greater amount of shade, and higher water holding capacity of the soils probably all contributed to the higher soil moisture in the spruce stringers. However, it should be noted that the summer this data was collected was wet and during dry summers when fires generally occur the difference in soil moisture between the burned area and spruce stringer would likely be much greater.

The higher soil moisture in the spruce stringers were likely responsible for the resistance to fire detected in the stands. The organic layer is an important fuel in boreal forest and it is likely that this layer remains to wet to burn during most years. The high soil moisture promotes vegetation with high water content in the foliage which tend to be less flammable than more sclerophyllic vegetation.

Discussion & conclusions

Analysis of vegetation data showed that stand composition within stringers is homogeneous and there is little difference among stand composition for the six stringers studied. On the other hand, stand composition of the adjacent forest was much more heterogeneous, both within and among the sites. Fire history for each of the adjacent forest was the same with stand-replacing fires occurring around 1930 and 1865. Fire history among the spruce stringers was also the same with all stringers having escaped fire for at least 200 years. The obvious difference in fire history between the spruce stringers and the adjacent forest suggested a higher resistance to fire. The primary reason for this appears to be their locations in small topographic depressions where groundwater surfaces as springs and seeps. This microrelief maintains high soil moisture and promotes growth of less flammable vegetation that also acts as a better source of shade for dead fuels on the forest floor. These combined effects create a fuel complex that has lower flammability than adjacent forest. This study shows microrelief in Alaskan boreal forest can have a dramatic impact on fire regimes and south facing slopes have a relatively high fire frequency that maintains early stages of succession (i.e. paper birch) indefinitely.

Reference: Yarie, J. 1981. Forest fire cycles and life tables: a case study from interior Alaska. Canadian Journal of Forest Research. 11(3): 554-562.

Introduction

Stand age was sampled over a large area in eastern interior Alaska and used to estimate fire frequency for the entire study area and sub-divisions stratified by three stand categories defined by dominant overstory species (black spruce, white spruce, and hardwoods). Additionally two age class distribution models (negative exponential and Weibull distributions) were evaluated.

Methods

The study area, located in the Porcupine and Upper Yukon River drainages, occupies 3.6 million hectares. Most of the study area falls within two physiographic regions: the Yukon Flats, an area of extensive lowlands, and the Porcupine Plateau, characterized by hilly terrain with gentle slopes. A total of 371 stands were sampled and 3-5 basal cross sections were collected from each stand. Each cross section was aged and the oldest sample was used to represent stand age. Stands were grouped into 10-year age classes up to 100 years, and 20-year age classes between 100 and 200 years. Stands older than 200 years were uncommon and disregarded. The 0-20 year age cohort was excluded from analysis to avoid potential influences of fire suppression. Stand age distribution was used to construct life tables and survivorship curves. The negative exponential and Weibull models were used to describe the survivorship curves. Life table analysis can only be conducted if three parameters are constant: population, renewal rate, and mortality rate. Based on paleoecological evidence indicating Alaska's climate and forest composition have been stable since 7000-6000 BP (Ager 1971, Lichtife.S 1973), these criteria were deemed to be sufficiently satisfied.

Results

For the study area and two of the three stand categories (hardwoods and black spruce) age class distribution exhibited a negative exponential distribution. Age class distribution for white spruce more closely resembled the Weibull distribution. Based on the Kolmogorov-Smirnov (K-S) statistic, both age-class distribution models were significant at the 0.05 level for the goodness of fit test for all stand categories and neither model was rejected. Fire cycle estimates produced by each model varied widely for some stand categories (table 16). Though fire cycle data from both models were presented, discussion, conclusions, and further analysis were primarily based on estimates from the Weibull model. The Weibull model was chosen over the negative exponential model because it more realistically incorporated changes in probability of burning that likely occurred over the wide range of stand categories and seral stages within the study area. The fire cycle for the entire study area was 43 years. Fire cycle was lowest in hardwoods (26 years), followed closely by black spruce (36 years). The fire cycle for white spruce (113 years) was much longer. If it is assumed that fire suppression has had no effect on fire occurrence and age classes under 20 years are included, the fire cycle was higher for all vegetation types except white spruce (table 16). The Weibull distribution shape parameter for the entire study area and hardwoods was one. It was lower (0.87) for black spruce and higher (2.36) for white spruce.

Age class data from the suppression period was excluded from life table analysis because fire cycle data indicated fire suppression had influenced mortality rate and had possibly shifted populations. Thus, at least one criteria of life table analysis had been violated. Life table analysis suggested there was little difference between the life expectancy of a stand (48 years) and the fire cycle (43 years) for the entire study area. Similar trends were observed for black spruce (43 vs. 36 years) and hardwoods (30 vs. 26 years). The opposite trend was observed for white spruce where fire cycle (113 years) was greater than life expectancy (105 years). Mortality rate data varied for each vegetation type (figure 24). Mortality rate of white spruce increased with time, and decreased over time for black spruce. For the entire study area and hardwoods, it remained relatively stable over time. The life expectancy for each stand over time continually decreased for all vegetation types except for black spruce. For black spruce, the life expectancy increased with age before declining in the 70-80 year age class.

Discussion & Conclusions

Fire cycle data suggested that fire regimes vary among stand categories. Fire frequency was higher for hardwoods and black spruce than white spruce. Fire cycles presented in previous research (Johnson 1979, Heinselman 1981) for these stand categories was generally higher than fire cycles presented here. The shape parameter data indicated how probability of burning changed with age. Hardwoods had an equal probability of burning over time while it steadily increased over time for white spruce. Probability of burning appeared to decrease slightly over time for black spruce, possibly a result of cooler and wetter conditions noted in mature stands of black spruce (Viereck 1973). The higher fire cycles calculated when age classes from the suppression period were included suggested fire suppression influenced age distributions and fire frequency within the study area. This result corresponded with other fire history studies that assessed the influence of fire suppression on fire regimes in boreal regions (Zackrisson 1977).

When compared with fire cycle calculations, the life expectancy data from life table analysis suggested that with the exception of white spruce, the original cohort should still be alive at the end of one fire cycle. The life expectancy data for white spruce suggested a small portion of stands will burn more than once during the fire cycle. Mortality rates correlated with shape parameter data. The declining mortality rates in black spruce correlated with the negative shape parameter indicating probability of burning decreased with age until the 70-80 year age class.

Tables and figures

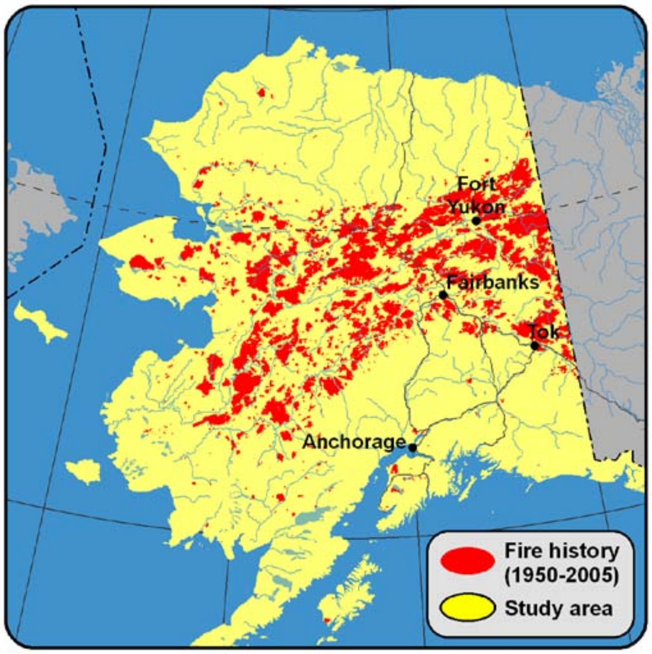

Figure 1. Study area (Barney 1971).

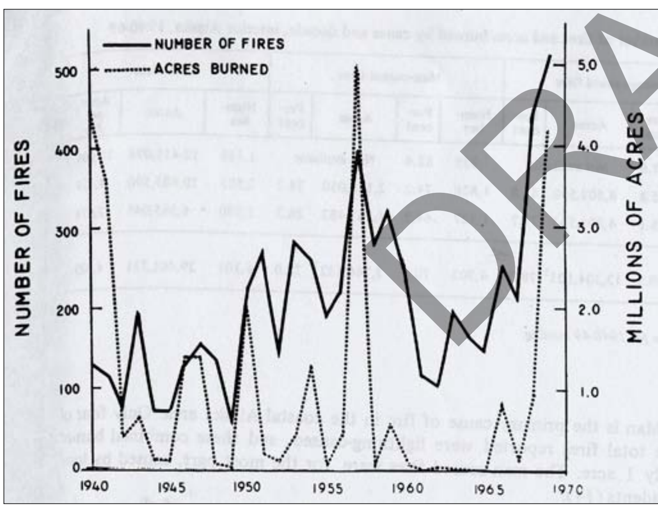

Figure 2. From Barney 1971, p. 53. Number of fires and area burned by year, interior Alaska, 1940-69.

Table 1. Number of fires and area burned by cause and decade, interior Alaska, 1940-69*.

Lightning-caused fires

Decade	Number	Percent	Hectares	Percent
1940-49	200	17.6	n/a	n/a
1950-59	745	25.8	3.5	61.8
1960-96	853	35.1	1.9	71.7
1940-69	1798	29.5	5.4	78.0

Human-caused fires

Decade	Number	Percent	Hectares	Percent
1940-49	938	82.4	n/a	n/a
1950-59	1838	74.2	0.9	38.2
1960-96	1527	64.9	0.6	28.3
1940-69	4303	70.5	1.5	22.0

Total

Decade	Number	Hectares	Mean fire size[†]
1940-49	1138	5.0	4413
1950-59	2583	4.3	1674
1960-96	2380	2.6	1082
1940-69	6101	11.7	1954

*Adapted from table 1 (p. 54) in Barney 1971.
[†]Hectares.

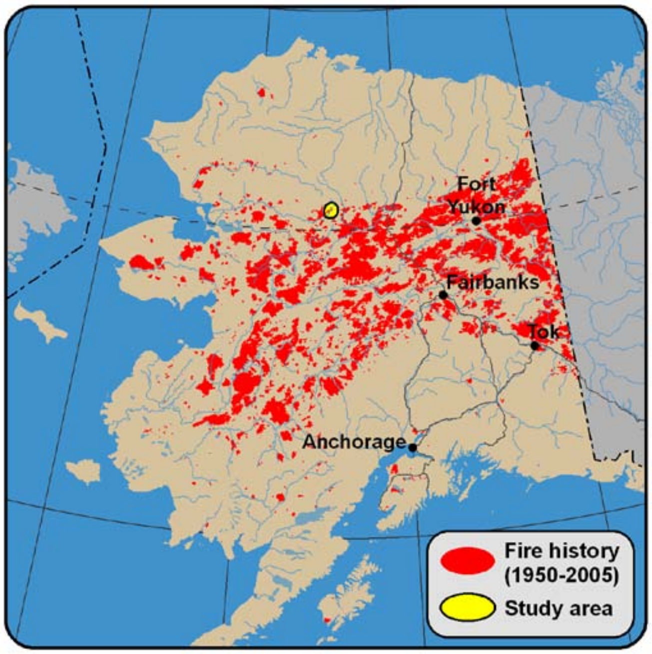

Figure 3. Study area (Christiansen 1988).

Table 2. Plot characteristics*.

Plot	B1	B2	B3	W1	W2	W3	W4
Trees in plot	163	108	75	64	71	144	74
% black spruce	87	100	100	15	34	47	37
% white spruce	13	0	0	85	66	53	63
Fire date†	1891	1891	1913	1891	1891	1891	1913
Fire survivors	2	14	0	0	4	15	3
Years after fire to establish 75% of trees	30	50	35	80	60	50	25
Years since last seedling established	15	0	30	0	0	20	25

*Adapted from table 1 (p. 32) in Christiansen 1988.
†Determined from fire scar data.

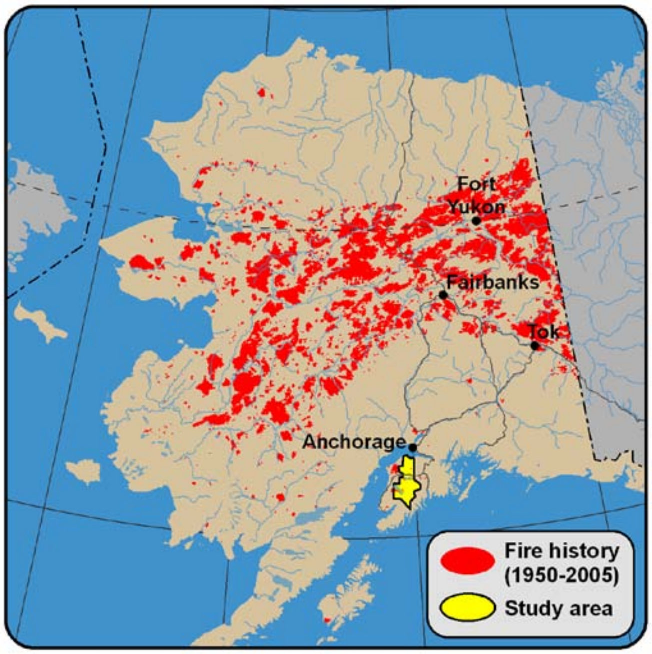

Figure 4. Study area (De Volder 1999).

Table 3. Natural fire regime data for the Kenai National Wildlife Refuge.

Topographic region	*mFRI**	Fire cycle†
All	89 ± 43	42
Mountainous	132 ± 52	n/a
Rainshadow	88 ± 41	n/a
Lakes	69 ± 36	n/a

*Reported in years, mean ± 1 SD. †Years.

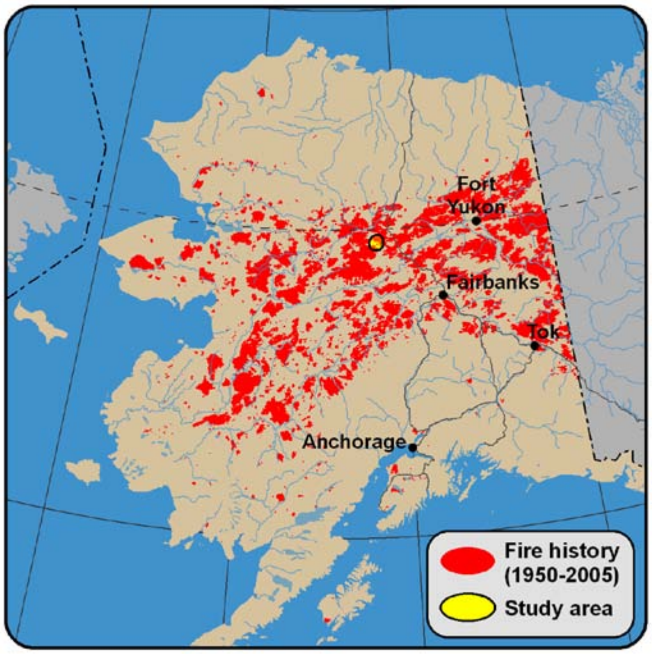

Figure 5. Study area (Earle et al. 1996).

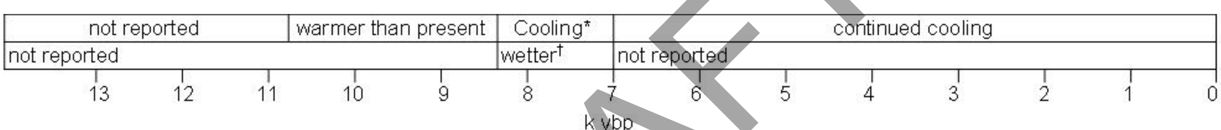

Figure 6. Climate reconstruction described relative to present day climate (Hu et al 1993, Anderson and Brubaker 1994). *Effective moisture increased, this was due to cooling temperatures and/or increased precipitation

Table 4. Relative changes in vegetation and fire regimes for Sithylemenkat Lake based on charcoal area for mid-size classes.

Vegetation type	Time period*	Fire regime†
Shrub birch tundra	14 000-10 000 BP	- -
White spruce-balsam poplar	10 500-9 000 BP	Unchanged
Not reported	9000-8000 BP	Unchanged
Alder becomes common	8000-6500 BP	Declined
Black spruce	6500-0 BP	Increased slightly

*Un-calibrated years. †Relative to previous period. Fire during the birch shrub tundra period was an important ecosystem process as indicated by a number of large charcoal peaks during this time.

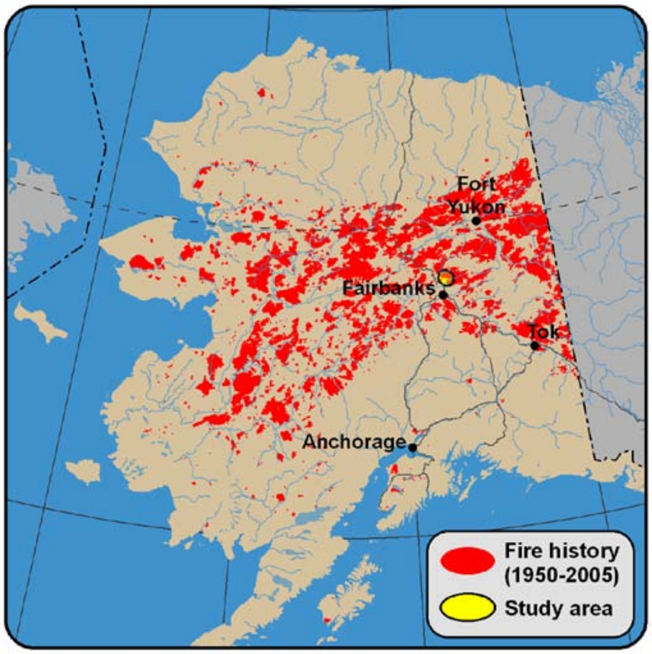

Figure 7. Study area (Fastie et al. 2002).

Table 5. Natural fire regime data for Caribou-Poker Creek Research Watershed.

Vegetation type	Sub-basin	Time period	*FRI**
Black spruce	C4	1750-1900	146
White spruce	C4	1750-2003	> 250†
Black spruce	P6	1800-1900	102

*Years. †May be influenced by 20th century fire suppression policies.

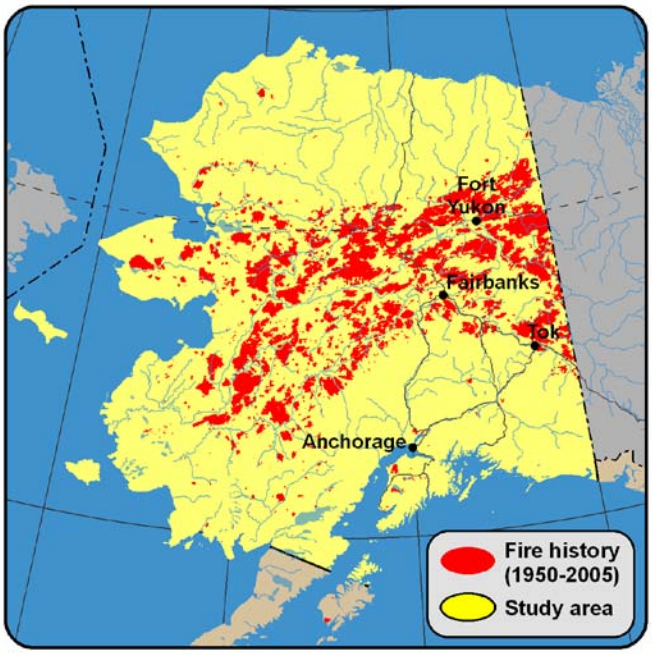

Figure 8. Study area (Gabriel and Tande 1983).

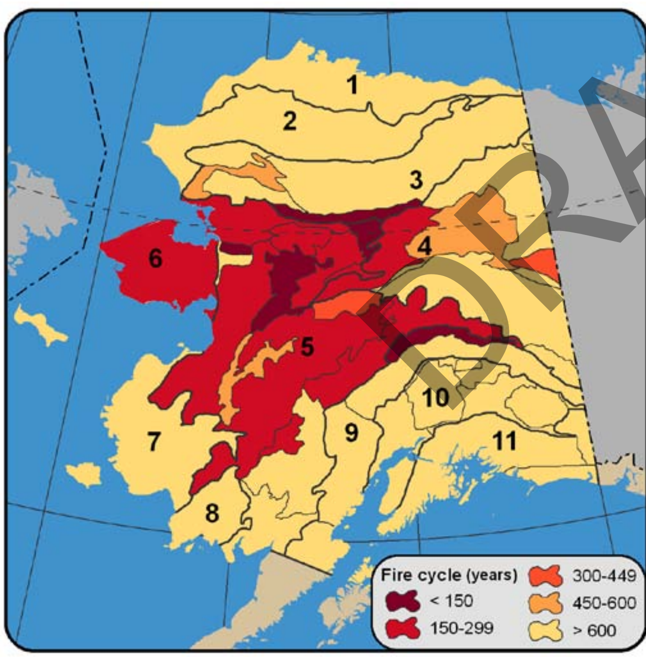

Figure 9. Fire cycle for physiographic sections. Numbers indicate physiographic provinces on table 6.

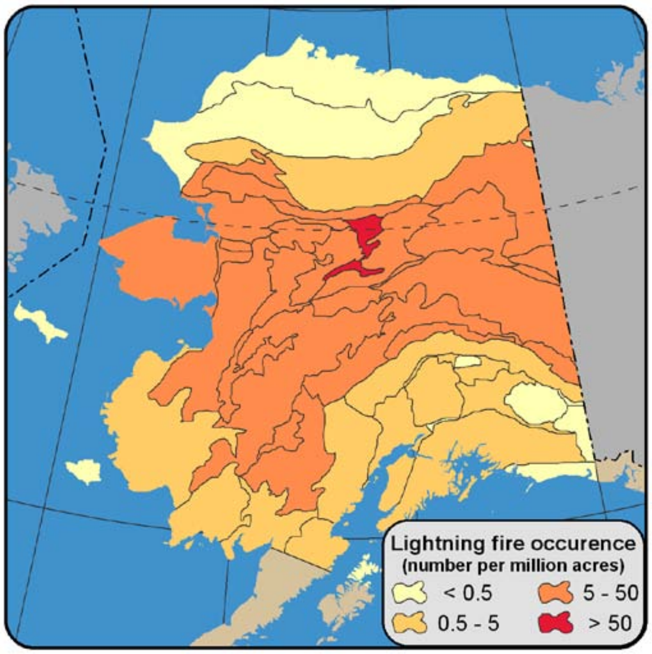

Figure 10. Occurrence of lightning caused fires per million acres for physiographic sections.

Table 6. Fire regime for physiological provinces in Alaska.

#*	Province	Fire cycle†	#*	Province	Fire cycle†
1	Arctic Coastal Plain	>1000	7	Bering Shelf	>1000
2	Arctic Foothills	>1000	8	Ahklum	>1000
3	Arctic Mountains	966	9	Alaska-Aleutian	533
4	Northern Plateaus	551	10	Coastal Trough	>1000
5	Western Alaska	140	11	Pacific Border Ranges	>1000
6	Seward Peninsula	263			

*Numbers to left of province names correspond with numbers in figure 9. †Years, calculations based on data presented in Gabriel & Tande (1983).

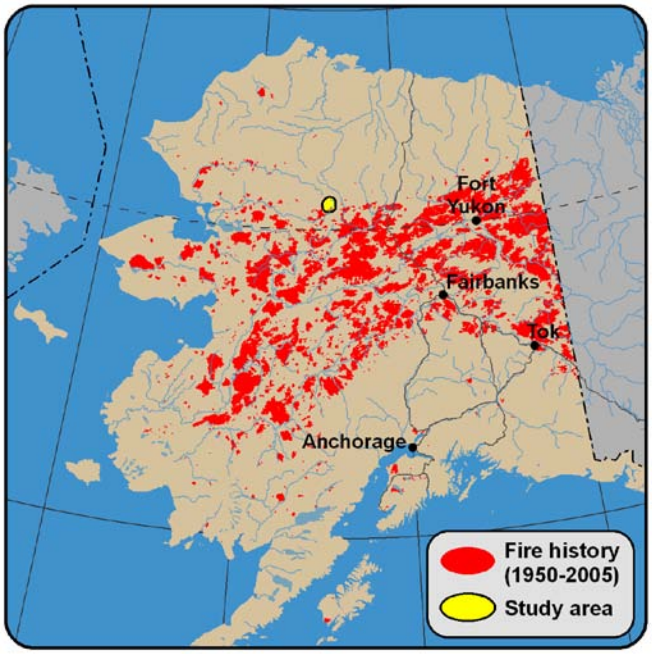

Figure 11. Study area (Goldstein 1981).

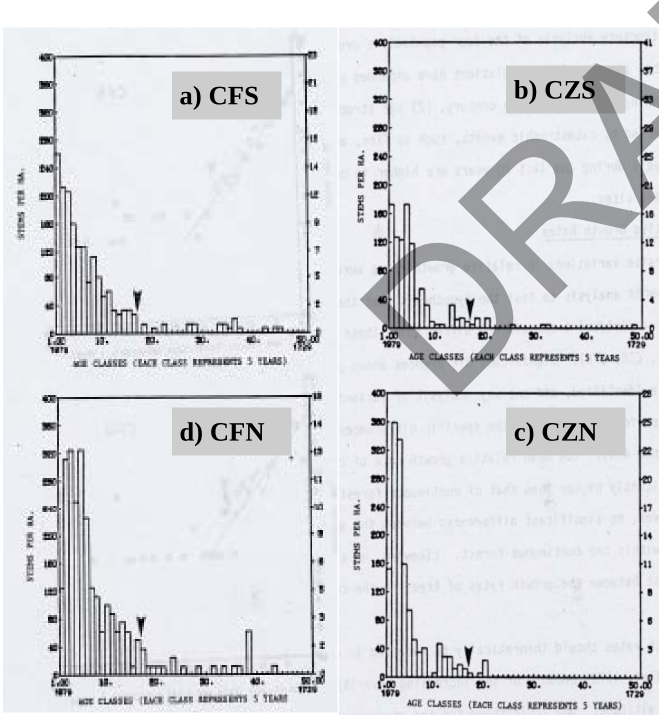

Figure 12. From Goldstein 1981, p. 37 & 41. The age structure of white spruce populations growing on north and south facing slopes. The left ordinate describes stems per ha and the right ordinate describes percent of total trees.

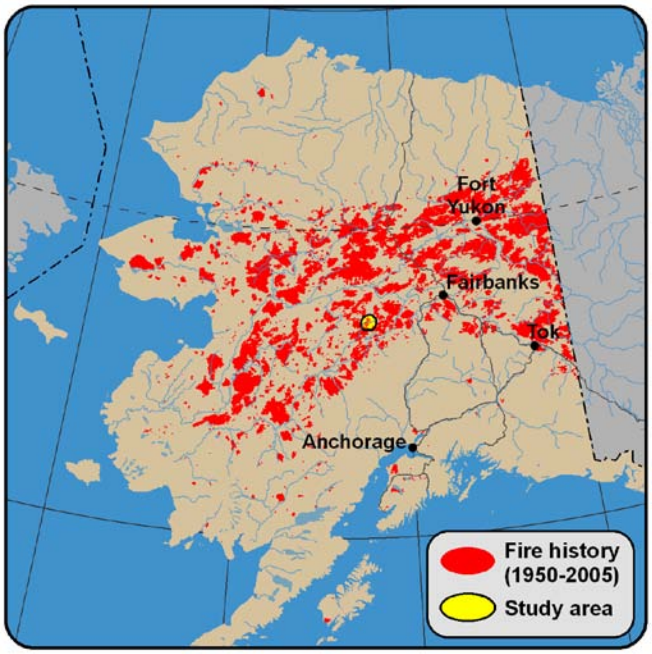

Figure 13. Study area (Hu et al. 1993).

Table 6. Relative changes of fire regimes for vegetation periods at Wein Lake based on macroscopic charcoal, pollen, and plant macrofossil data.

Vegetation type	Time period*	Fire regime†	Soils
Shrub birch tundra	12 000-10 500 BP	Sparse	Sparsely vegetated entisols
Balsam poplar gallery forest	10 500-9 500 BP	None	Forest inceptisols
White spruce, paper birch woodland	9500-6500 BP	Intermittent	Forest inceptisols
Black spruce closed forest	6500-0 BP	Frequent	Forest inceptisols

*Un-calibrated years. †Described by occurrence of charcoal in the sediment record.

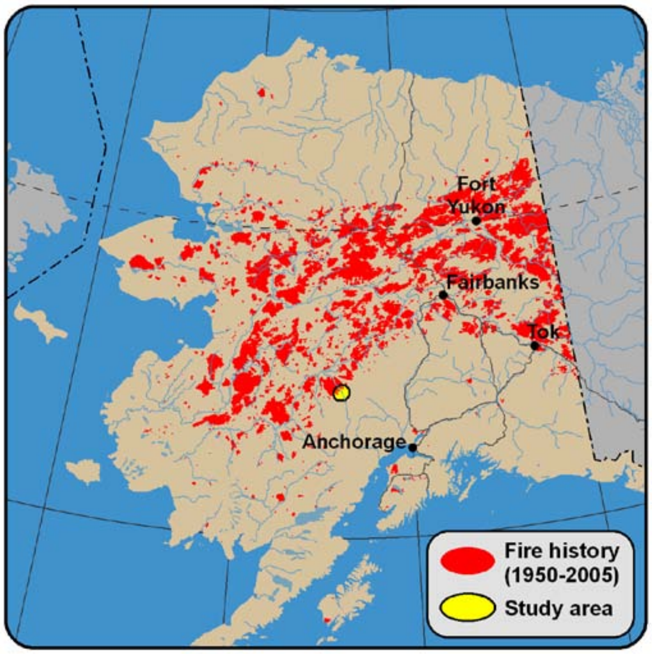

Figure 14. Study area (Hu et al. 1996).

Table 7. Relative changes in vegetation and fire regimes for Sithylemenkat Lake based on charcoal area for mid-size classes.

Vegetation type	Time period*	Fire regime†
Shrub birch tundra & balsam poplar gallery forest	11 000-8 000 BP	Intermediate
White spruce-paper birch forest-tundra	8000-4000 BP	Infrequent
Black spruce closed boreal forest	4000-0 BP	Frequent

*Un-calibrated. †Indicates fire frequency relative to previous periods.

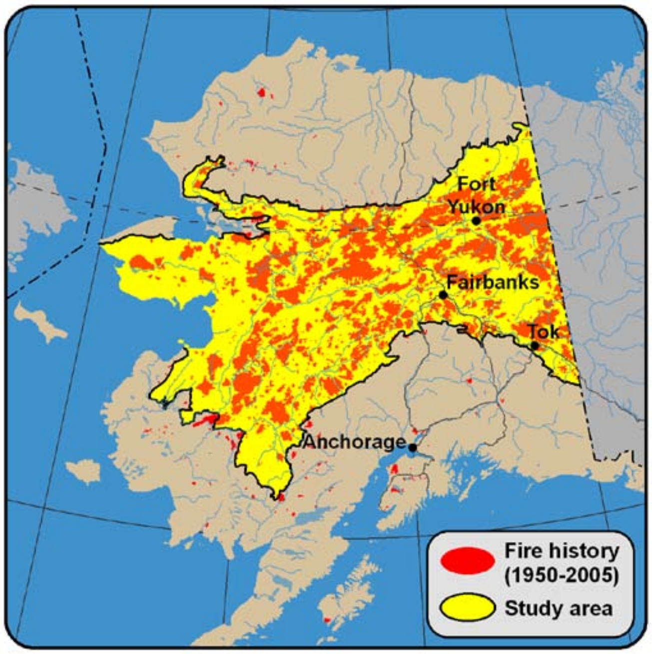

Figure 15. Study area (Kasischke et al. 2002).

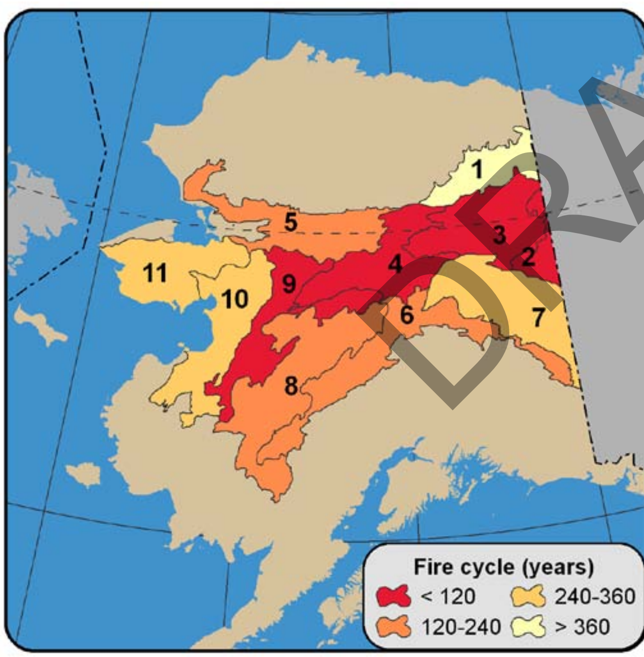

Figure 16. Fire cycle by ecoregion for interior Alaska. Ecoregion numbers correspond with names in table 9.

Table 8. Fire cycle as a function of elevation class.

Eco-region*	< 800	Fire cycle (years) by elevation class (m) 0-199	200-399	400-599	600-799
1	484	n/a	224	314	949
2	150	318	118	131	210
3	107	149	87	93	76
4	179	205	132	175	322
5	229	207	206	763	2172
6	244	285	170	311	294
7	588	1322	443	329	454
8	281	195	331	571	1536
9	174	171	230	n/a	n/a
All†	200	193	166	237	427

*Correspond with names listed in table 9. †Excludes coastal ecoregions.

Table 9. Ecoregions of interior Alaska.

1	Davidson Mountains	7	Yukon-Tanana Uplands
2	North Ogilvie Mountains	8	Kuskokwim Mountains
3	Yukon-Old Crow Basin	9	Nulato Hills
4	Ray Mountains	10	Seward Peninsula
5	Kobuk Ridges and Valleys	11	Yukon River Lowlands
6	Tanana-Kuskokwim Lowlands		

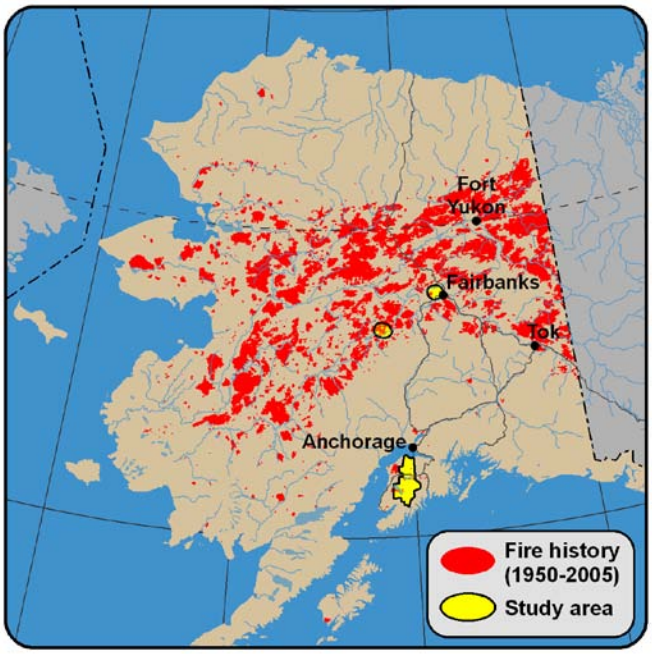

Figure 17. Study area (Lynch et al. 2002).

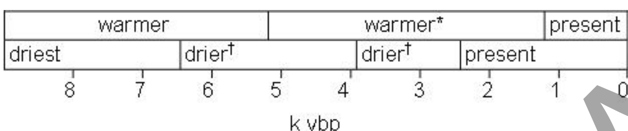

Figure 18. Climate reconstruction described relative to present day climate (Ager 1971, Anderson and Brubaker 1994, Edwards and Barker 1994, Bigelow 1997, Hu et al. 1998, Abbott et al. 2000, Finney et al. 2000. *Declining temperature. †Wetter than previous period.

Table 10. CHARs* by site and time period.

Location	1000-150 BP	150-0 BP
Interior Alaska		
Dune Lake	0.028 ± 0.03	0.073 ± 0.09
Deuce Lake	0.016 ± 0.03	0.027 ± 0.06
Kenai Peninsula		
Rock Lake	0.013 ± 0.02	0.053 ± 0.07
Portage Lake	0.020 ± 0.03	0.110 ± 0.10
Arrow Lake	n/a	0.118 ± 0.14

*$mm^2/cm^2/yr^1$ ± SD

Table 11. Fire regime data for Dune Lake.

Vegetation type	Time period*	*mFRI*†	c parameter‡
Birch-spruce	9000-8000 BP	n/a	n/a
Birch-spruce-alder	8000-5500 BP	n/a	n/a
Black spruce	5500-2400 BP	97± 68	1.6
Black spruce	2400-0 BP	198± 90	2.8

*Calibrated years. †Years ± 2 SD. ‡Weibull shape parameter

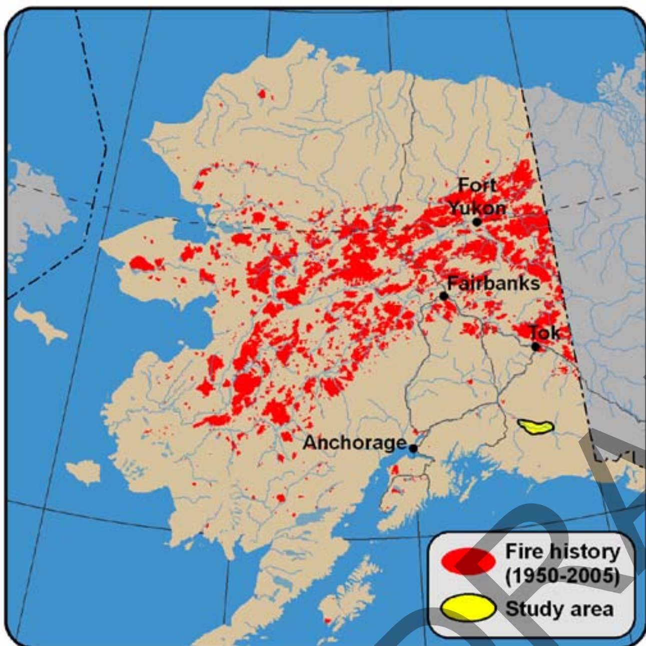

Figure 19. Study area (Lynch et al. 2004).

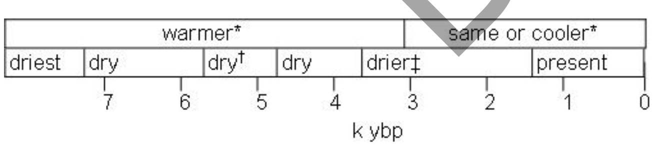

Figure 20. Climate reconstruction described relative to present day climate for Chokasna and Moose lakes. *Calkin 1988, Wiles et al 2002. †But wetter than latter periods. ‡Wetter than previous period.

Table 12. Fire regime data.

Vegetation type	Time period*	mFRI†	CHAR peaks‡
Moose Lake			
Mixed boreal forest	> 5800 BP	> 500	> 1
Mixed boreal forest	5800-5000 BP	190 ± 20	> 0.018
Mixed boreal forest	5000-3800 BP	> 500	< 0.01
Mixed boreal forest	3800-0 BP	210 ± 80	> 0.018
Chokasna Lake			
Mixed boreal forest	> 6600 BP	> 500	> 0.085
Mixed boreal forest	6600-5600 BP	> 500	< 0.01
Mixed boreal forest	5600-4500 BP	220 ± 60	> 0.085
Mixed boreal forest	4500-3100 BP	> 500	< 0.01
Mixed boreal forest	3100-2000 BP	295 ± 20	~ 0.16
Black spruce forest	2000-0 BP	150 ± 80	> 0.16

*k ybp. †Years ± SD. ‡$mm^2/cm^2/yr^1$.

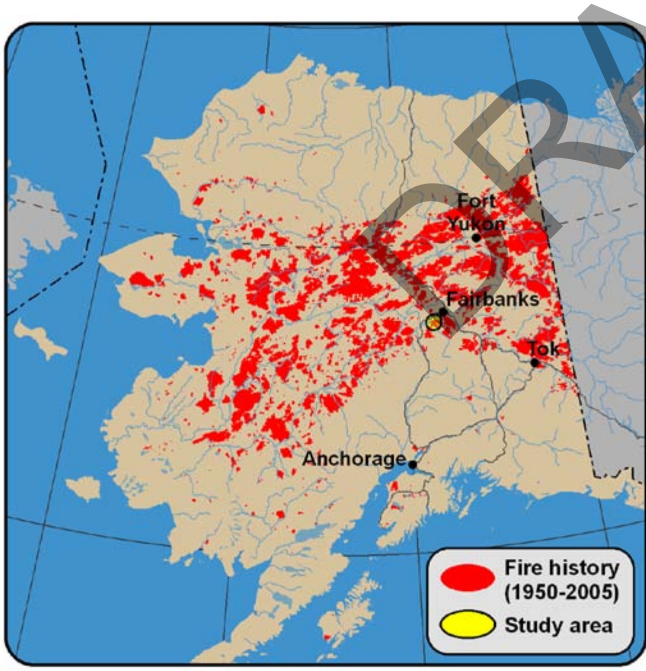

Figure 21. Study area (Mann et al. 1995).

Table 13. Fire regime for the Tanana River floodplain.

Vegetation type	FRI (range)*
Spruce forest	70—110

*Years.

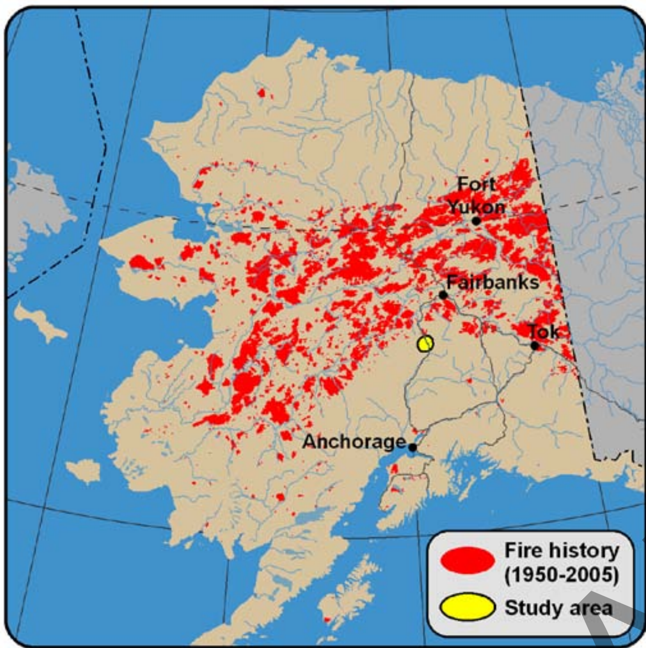

Figure 22. Study area (Mann and Plug 1999).

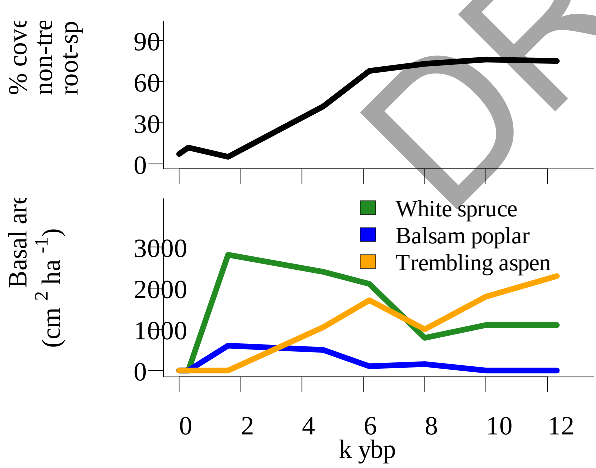

Figure 23-24. Vegetation properties: chronosequence for the Riley Creek floodplain, Denali National Park, Alaska.

Table 14. Fire regime for the Riley Creek floodplain, Denali National Park, Alaska.

Vegetation type	FRI (range)*
Trembling aspen	40—60

*Years.

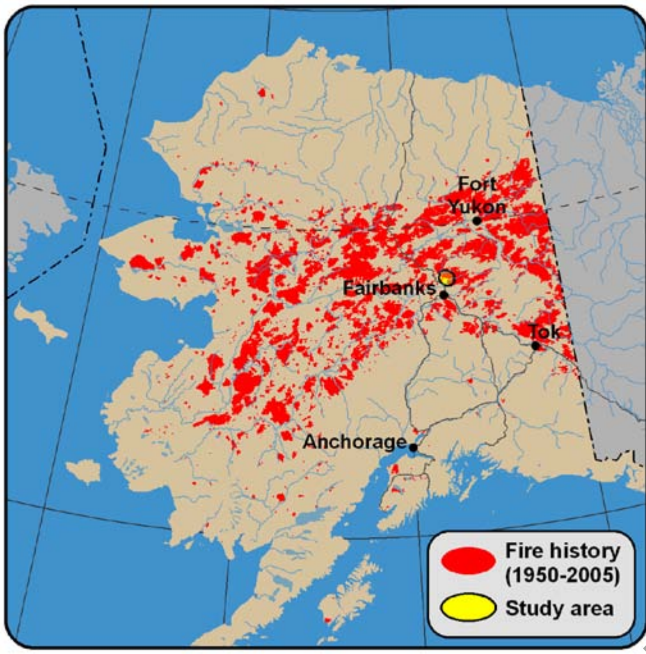

Figure 24. Study area (Quirk and Sykes 1971).

Table 15. Fire regime data for boreal forest in the Caribou-Poker Creeks Research Watershed, Alaska.

Vegetation type	Time period	FRI*
White spruce	1762-1970	> 200
Mixed boreal	1865-1970	65

*Reported in years.

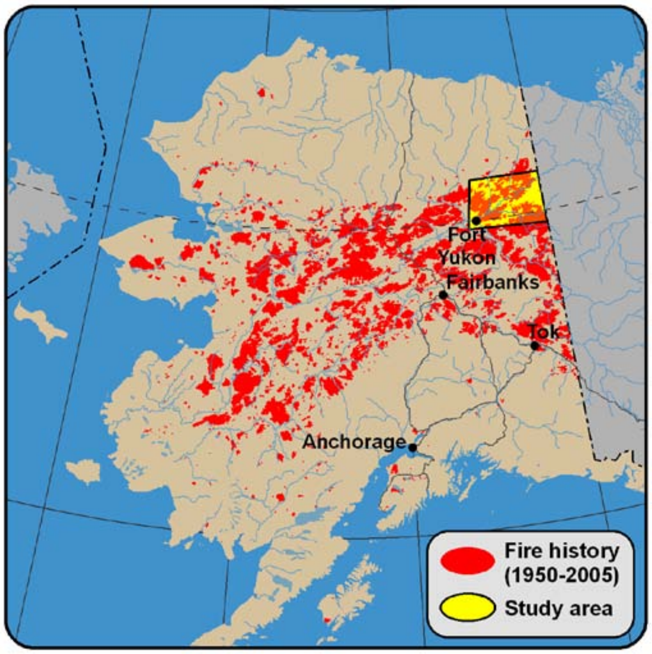

Figure 25. Study area (Yarie 1981).

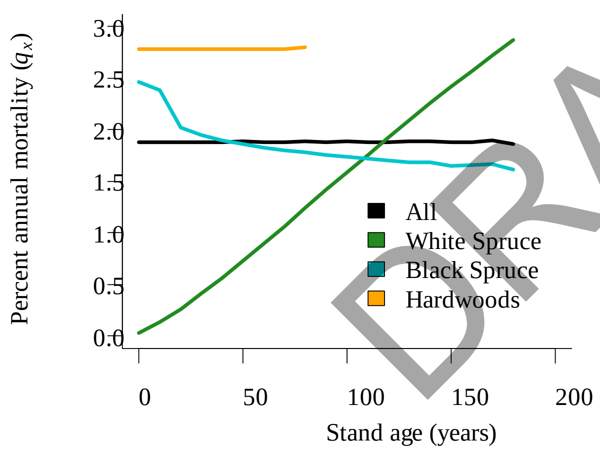

Figure 26. Estimated annual mortality rate relative to stand age for common vegetation types of the Porcupine and Upper Yukon watersheds, Alaska.

Table 16. Fire regime data for the Porcupine and Upper Yukon watersheds, Alaska.

		Fire cycle†		
Vegetation type	Shape parameter*	Neg. exp. distribution	Weibull distribution	With fire sup. data‡
All	1.00	43	43	93
White spruce	2.36	53	113	113
Black spruce	0.87	45	36	100
Hardwoods	1.00	20	26	60

*For the Weibull distribution. †Reported in years. ‡Based on the Weibull distribution